2008

The role of aquatic vegetation in Iowa lakes

Megan Ernst
Iowa State University

The role of aquatic vegetation in Iowa lakes

by

Megan Ann Ernst

A thesis submitted to the graduate faculty

in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Major: Fisheries Biology

Program of Study Committee:
Joseph E. Morris, Major Professor
Michael C. Quist
John A. Downing

Iowa State University

Ames, Iowa

2008

TABLE OF CONTENTS

LIST OF TABLES	iii
LIST OF FIGURES	vi
CHAPTER 1. GENERAL INTRODUCTION	1
Objectives	5
Thesis Organization	6
References	6
CHAPTER 2. ABIOTIC FACTORS INFLUENCE OF AQUATIC VEGETATION ABUNDANCE IN IOWA LAKES	12
Abstract	12
Introduction	13
Study Area	17
Methods	18
Results	20
Discussion	24
References	28
CHAPTER 3. LITTORAL INFLUENCES ON ZOOPLANKTON POPULATIONS AND JUVENILE BLUEGILLS IN IOWA LAKES	57
Abstract	57
Introduction	58
Study Area	62
Methods	63
Results	66
Discussion	68
References	74
CHAPTER 4. GENERAL CONCLUSIONS	90
References	94
ACKNOWLEDGMENTS	97

LIST OF TABLES

CHAPTER 2.

Table 1. Summary information for the 13 Iowa study lakes in 2007. Information summarized is: Lake, county, mean depth (m), lake size (ha), density of grass carp *Ctenopharyngodon idella* (fish/ha). 35

Table 2. Summary statistics for environmental parameters measured during the sampling of emergent/floating and submerged aquatic vegetation in 13 Iowa lakes from May 2007 to September 2007. 36

Table 3. Summary correlations (r^2) of simple linear regressions between arcsine(square root) transformed emergent/floating vegetation abundance and limnetic environmental parameters in 13 Iowa Lakes in 2007. Statistical significant relationships between emergent/floating vegetation abundance and physical-chemical parameters are determined by ($P \leq 0.05$). Significant values are in bold. 37

Table 4. Summary correlations (r^2) of simple linear regressions between arcsine(square root) transformed submerged vegetation abundance and physical-chemical parameters in 13 Iowa Lakes in 2007. Statistical significant relationships between submerged vegetation abundance and environmental parameters are determined by ($P \leq 0.05$). Significant values are in bold. 37

Table 5. Emergent/floating aquatic vegetation summary statistics for the 13 Iowa lakes during the months of May 2007 to September 2007. Information summarized is: number of samples, mean emergent/floating vegetation abundance and standard error (SE), Shannon Index of diversity, species present, number of times that species was identified (N), individual species abundance and SE, and the percent each species contributes to the overall emergent/floating vegetation abundance. Abundance index (%) <0.1= trace (tr). 38

Table 6. Submerged aquatic vegetation summary statistics for the 13 Iowa lakes during the months of May 2007 to September 2007. Information summarized is: number of samples, mean submerged vegetation abundance and standard error (SE), Shannon Index of diversity, species present, number of times that species was identified (N), individual species abundance and SE, and the percent each species contributes to the overall submerged vegetation abundance. Abundance index (%) <0.1= trace (tr). 46

Table 7. Summary of nMDS r^2 and P-value of the physical-chemical parameters in 13 Iowa Lakes in 2007. P-value is based on 1000 permutations. Statistically significant relationships of vegetation abundance (i.e., emergent/floating and submerged) and physical-chemical parameters are determined by ($P \leq 0.05$). Significant values are in bold. 52

CHAPTER 3.

Table 1. Summary information for the three Iowa study lakes: lake, county, mean depth (m), lake size(ha), and density of grass carp (fish/ha). 83

Table 2. Mean ± SE statistics for environmental parameters measured during the sampling of emergent/floating and submerged aquatic vegetation in three Iowa lakes from May 2007 to September 2007. 83

Table 3. Mean ± SE seasonal zooplankton densities in three Iowa lakes in 2007. Samples were collected by a water pump in the limnetic zone, non-vegetated littoral zone (Open), and vegetated littoral zone (Veg). In addition, a vegetated littoral sample was collected using a box sampler (Box). Levels of densities less than one individual per/liter is labeled as trace (tr). 84

Table 4. Summary of one-way analysis of variance comparing zooplankton samples collected in the limentic, littoral vegetated (Veg), littoral non-vegetated (Open), and littoral vegetated box sample (Box) in three Iowa lakes in 2007. Significant relationship is determined by Bonferroni corrected P-value ≤0.003 (bold). 85

Table 5. Total stomach contents of juvenile (≤50 mm) bluegills from spring/summer and fall electrofishing at three Iowa lakes in 2007. 86

Table 6. Total stomach contents of juvenile (>50 mm) bluegills from spring/summer and fall electrofishing at three Iowa lakes in 2007. 87

Table 7. Mean ± SE (P-value) linear indices[1] of food selection of juvenile (≤50 mm) bluegill from spring and fall electrofishing at Ahquabi, Red Haw, and Wapello, Iowa, 2007. Significant relationship is determined by a P-value ≤.05 (bold). 88

Table 8. Mean ± SE (P-value) linear indices[1] of food selection of juvenile (≥50 mm) bluegill from spring and fall electrofishing at Ahquabi, Red Haw, and Wapello, Iowa, 2007. Significant relationship is determined by a P-value ≤.05 (bold). 89

LIST OF FIGURES

CHAPTER 2.

Figure 1. Summary of simple linear regression plots between arcsine (square root) transformed emergent/floating vegetation abundance and mean physical-chemical parameters (i.e, alkalinity, chlorophyll *a*, hardness, and temperature) in 13 Iowa lakes in 2007. Statistically significant relationships between emergent/floating vegetation abundance and mean environmental parameters are determined by ($P \leq 0.05$). 53

Figure 2. Summary of simple linear regression plots between arcsine (square root) transformed submerged vegetation abundance and mean physical-chemical parameters (i.e., chlorophyll a, Secchi-depth, total Kjeldahl nitrogen, and total suspended solids) in 13 Iowa lakes in 2007. Statistically significant relationships between submerged vegetation abundance and mean environmental parameters are determined by ($P \leq 0.05$). 54

Figure 3. Non-metric multidimensional scaling (NMDS) plot illustrating the strength and relationship among physical-chemical parameters (vectors), lakes (underlined), and emergent/floating aquatic vegetation (gray) in 13 Iowa lakes in 2007. Plant vegetation codes are located in Table 5. 55

Figure 4. Non-metric multidimensional scaling (NMDS) plot illustrating the strength and relationship among environmental parameters (vectors), lakes (underlined), and submerged aquatic vegetation (gray) in 13 Iowa lakes in 2007. Plant vegetation codes are located in Table 6. 56

CHAPTER 3.

Figure 1. Location and aerial overview of three Iowa study lakes (e.g., Lake Ahquabi, Red Haw Lake, and Lake Wapello). 82

CHAPTER 1. GENERAL INTRODUCTION

The role of aquatic vegetation in fisheries management decisions has often been influenced by ongoing conflicts between a fishery and its users. For instance, aquatic vegetation has been viewed as a nuisance “weed” by limiting entry to water bodies, causing frustration among anglers, and making lakes unpleasant for recreational activities (i.e., slow growing sport fish through reduced feeding rates and predator forage efficiency; Crowder and Cooper 1982; Bettoli et al. 1993). In contrast, the importance of aquatic vegetation to the health of ecosystems has been documented by numerous studies (Crowder and Cooper 1979; Savino and Stein 1982; Durocher et al 1984; Bettoli et al. 1993).

In regions such as the Midwest that have a highly altered landscape, rural and urban influences can contribute excess nutrients and soil loss creating aquatic systems inundated with vegetation and plagued with algal blooms. These anthropogenic changes cause eutrophication of lakes, altering aquatic vegetation species and abundance (Scheffer et al. 2002; Egertson et al. 2004)

Aquatic vegetation plays a vital role in maintaining the overall integrity of aquatic ecosystems. Vegetation stabilizes aquatic ecosystems by reducing nutrient concentrations (van Donk et al. 1989) and shoreline erosion, providing food and habitat for aquatic fauna, and increasing water clarity, producing oxygen, reducing shore erosion (Canfield et al. 1984; Timms and Moss 1984; Jeppesen et al. 1990; Scheffer et al. 1993; Meijer et al. 1994; Moss et al. 1994; Egertson et al. 2004). Aquatic vegetation abundance is influenced by factors such as irradiance, temperature, water chemistry (nitrogenous and phosphorus nutrients), wave action, lake size, and catchment basin morphology (Gasith and Hoyer 1998).

In addition to the already perceived over-abundance of native aquatic plants in many lakes, the introduction of non-native plants has further disrupted the natural balance in aquatic systems throughout the United States. Iowa's exotics — Eurasian watermilfoil *Myriophyllum spicatum*, brittle naiad *Najas minor*, and curly pondweed *Potamogeton crispus*— are spreading rapidly, out-competing native plants, and altering fish and wildlife habitat and behavior (AERF 2005). Exotic species have invaded healthy and degraded ecosystems creating monoculture stands with dense canopies and decreasing the ability of native vegetation to survive. Excessive vegetative growth, unchecked by native populations, is responsible for deterioration of fish and wildlife habitat, wetlands, and water quality, reduction in property value, impediment of recreational activities and commercial navigation, and blockage of water supply intakes.

In recent decades, pollutant inputs from agricultural and urban landscapes have further increased and degraded the water quality of rivers, lakes, and coastal oceans (Carpenter et al. 1998). Water quality degradation is the loss of natural systems, their component species, and the amenities they provide (U.S. EPA 1996; Postel and Carpenter 1997). Eutrophication is the most common impairment of surface waters in the United States, stemming from excessive nutrient loading of phosphorus and nitrogen (U.S. EPA 1996). Eutrophication accounts for ~50% of all impaired lake areas in the United States (U.S. EPA 1996). Iowa's rich agricultural landscape (ca. 9.7 million hectares of tilled cropland; USDA 2004) further increases the already naturally-enriched nutrients of many of Iowa's lakes.

Although aquatic plants are sometimes perceived as all being the same in their role in aquatic systems, there are fundamental differences among plant types. For instance, submerged vegetation is important in stabilizing the clear water state in shallow, mesotrophic, and eutrophic lakes (Blindow and Hootsmans 1991; Simons et al. 1994; Perrow et al. 1997; Scheffer 2004). Charophytes (*Chara* spp.) have a strong positive effect on water transparency compared to other aquatic plants (Scheffer et al. 1993; van den Berg et al. 1994, 1998). Rooney et al. (2003) determined that submerged aquatic vegetation beds accumulate twice as much bulk sediment per unit area as the profundal zone. Phosphorus levels in these beds were only 1/6th of that in the profundal zone, losing >70% sedimented phosphorus after deposition, making aquatic vegetation a possible management tool to filter and protect water supplies from pollution.

In addition to the previously described physical effects of aquatic plants, their presence in eutrophic and mesotrophic lakes have a positive effect on zooplankton biomass and a negative effect on phytoplankton biomass (van Donk and van de Bund 2002). Submerged vegetation provides refuge for algae-eating zooplankton (e.g., cladocerans escapement from zooplankitvorous fish; Timms and Moss 1984). Aquatic vegetation also indirectly contributes to fish growth and recruitment by increasing and diversifying invertebrate communities as well as providing age-0 fish with protection from predation by reducing some predators' visibility and maneuverability (Savino and Stein 1982). Dense macrophytes can actually serve as refugees for large-bodied cladocerans escaping from predation from zooplanktivorous fish which is consistent with the refugee hypothesis for grazing zooplankton (Timms and Moss 1984; Stansfield et al. 1997).

Members of the Centrarchidae family are well known inhabitants of littoral zones of lakes. Centrarchids have been shown to use vegetated zones for protection against predation and use non-vegetated zones to maximize foraging efficiency on visible zooplankton (Werner et al. 1983; Werner and Hall 1988). Given the importance of vegetated habitats to centrarchids, major changes resulting from a reduction of vegetation abundance can occur (Bettoli et al. 1993). After a drastic macrophyte removal in Lake Conroe, Texas, algal biomass increased rapidly, cyanobacteria dominated summer blooms, water clarity decreased, and bluegill *Lepomis macrochirus* and major zooplankton taxa biomass decreased (Bettoli et al. 1993).

In addition to acting as a phosphorus sink, aquatic vegetation also decreases the availability of nitrogen for phytoplankton growth (Weisner et al.1994) that can affect higher trophic levels (e.g., zooplankton and fish; van Donk and van de Bund 2002). Campbell et al. (1985) and Bettoli et al. (1991) found that aquatic vegetation beds in Lake Conroe supported an abundant microcrustacean community, dominated by cladocerans and cyclopoid copepods, due to the increased number of interstitial spaces associated with the increased complexity of aquatic vegetation beds. Prior to vegetation removal in 1985, the total density of littoral invertebrate prey was more than $500L^{-1}$ and after vegetation removal, densities decreased to < $2L^{-1}$ (Campbell et al. 1985; Bettoli et al. 1991) resulting in population shifts of competing zooplanktivorous fish (i.e., inland *Menida beryllina* and brook silversides *Labidesthes sicculus*; Bettoli et al. 1991). This is only one of several publications that describe the important role of aquatic vegetation in aquatic ecosystems.

Given the complexity of the aquatic vegetation often found in lakes, there is no one long-term solution to their management although herbicides, grass carp *Ctenopharyngodon idella*, lake drawdowns, and mechanical methods have been used to manage the short term issues of aquatic vegetation. As previously noted, both the complete eradication and severe infestation of aquatic vegetation can be detrimental to fish populations. In light of these issues, the goal of modern fisheries management is to maintain intermediate abundance of aquatic vegetation whereby it is possible to optimize the lakes' resources (Wiley et al. 1984). The best solution to managing aquatic vegetation is a combination of preventative, physical, biological, and chemical control specific for that lake's environmental conditions and fishery needs.

OBJECTIVES

The goal of this study was to assess the interrelationship of aquatic vegetation and physical-chemical parameters, zooplankton densities, and food habits of juvenile bluegills in Iowa lakes. The first objective was to determine abiotic factors influencing on vegetation abundance. The second objective was to determine aquatic vegetation abundance and species influence on littoral zooplankton populations and food habitats of juvenile bluegills.

THESIS ORGANIZATION

This thesis has been organized into four chapters. Chapter one is a general introduction to my thesis research. Chapter two and chapter three are two manuscripts that will be submitted to *Journal of Aquatic Plant Management and North American Journal of Fisheries Management*, respectively. Chapter four contains general conclusions highlighting the results of my research.

REFERENCES

Aquatic Ecosystem Restoration Foundation (AERF). 2005. Management practices handbook for aquatic plant management in support of fish and wildlife habitat. Marietta, Georgia.

Bettoli, P.W., M.J. Maceina, R.L. Noble, and R.K. Betsill. 1993. Response of a reservoir fish community to aquatic vegetation removal. North American Journal of Fisheries Management 13:110-124.

Bettoli, P.W., J.E. Morris, and R.L. Noble. 1991. Changes in the abundance of two atherinid species after aquatic vegetation removal. Transactions of the American Fisheries Society 120: 90-97.

Blindow I. and M.J.M Hootsmans. 1991. Allelopathic effects from *Chara* spp. on two species of unicellular green algae. Pages 139-144 *in* Hootsmans, M.J.M. and J.E. Vermaat (editors), Macrophytes as a key to understanding changes caused by eutrophication in shallow freshwater ecosystems. Institute for Hydraulic and Environmental Engineering. Delft, Netherlands.

Campbell, J.M., J.E. Morris, and R.L. Noble. 1985. Spatial variability and community structure of littoral microcrustacea in Lake Conroe, Texas. Journal of the Texas Academy of Science 6: 247-257.

Canfield, D.E. Jr., J.V. Shireman, D.E. Colle, W.T. Haller, C.E. Watkins II, and M.J. Maceina. 1984. Prediction of chlorophyll *a* concentrations in Florida lakes: importance of aquatic macrophytes. Canadian Journal of Fisheries and Aquatic Sciences 41:487-501.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8:559-568.

Crowder, L.B. and W.E. Cooper. 1979. Structural complexity and fish-prey interactions in ponds: a point of view. Pages 1-10 *in* Johnson, D.L. and R.A. Stein (editors), Response of fish to habitat structure in standing water. North Central Division. American Fisheries Society Special Publications 6: 1-10.

Crowder, L.B. and W.E. Cooper. 1982. Habitat structural complexity and the interaction between bluegills and their prey. Ecology 63: 1802-1813.

Durocher, P.P., W.C. Provine, and J.E. Kraii. 1984. Relationship between abundance of largemouth bass and submerged vegetation in Texas reservoirs. North American Journal of Fisheries Management 4:84-88.

Egertson, C.J., J.A. Kopaska, and J.A. Downing. 2004. A century of change in macrophyte abundance and composition in response to agricultural eutrophication. Hydrobiologia 524:145-156.

Gasith, A. and M.V. Hoyer. 1998. Structuring role of macrophytes in lakes: changing influence along lake size and depth gradients. Pages 381-389 *in* Jeppesen, E., Søndergaard, M., Christoffersen K. (editors), The structuring role of submerged macrophytes in lakes. Springer, New York.

Jeppesen, E., J.P. Jensen, P. Kristensen, M. Søndergaard, E. Mortensen, O. Sortkjaer, and K. Olrik. 1990. Fish manipulation as a lake restoration tool in shallow, eutrophic, temperate lakes 2: Threshold levels, long-term stability and conclusions. Hydrobiologia 200/201: 219-227.

Meijer, M-L., E. Jeppesen, E. van Donk, B. Moss, M. Scheffer, E. Lammens, E. van Nes, J.A. van Berkum, G.J. de Jong, B.A. Faafeng, and J.P. Jensen. 1994. Long-term responses to fish-stock reduction in small shallow lakes: interpretation of five-year results of four biomanipulation cases in The Netherlands and Denmark. Hydrobiologia 275/276: 457-466.

Moss, B., S. McGowan, and L. Carvalho. 1994. Determination of phytoplankton crops by top-down and bottom-up mechanisms in a group of English lakes, the West Midland Meres. Limnology and Oceanography 39:1020-1029.

Perrow, M.R., M-L. Meijer, P. Dawidowicz, and H. Coops. 1997. Biomanipulation in shallow lakes: state of the art. Hydrobiologia 342/343:355-365.

Postel, S. and S.R. Carpenter. 1997. Freshwater ecosystem services. Pages 195-214 *in* G. Daily, (editor), Nature’s services. Island Press, Washington, D.C., USA.

Rooney, N., J. Kalff, and C. Habel. 2003. The role of submerged macrophyte beds in phosphorus and sediment accumulation in Lake Memphremagog, Quebec, Canada. Limnology and Oceanography 48:1927-1937.

Savino, J. and R.A. Stein. 1982. Predator-prey interaction between largemouth bass and bluegills as influenced by simulated, submersed vegetation. Transactions of the American Fisheries Society 111:255-266.

Scheffer, M. 2004. Ecology of shallow lakes. Chapman and Hall, London.

Scheffer, M., S. Carpenter, J.A. Foley, C. Folke and B. Walker. 2002. Catastrophic shifts in ecosystems. Nature 413:591-596.

Scheffer, M., S.H. Hosper, M.-L. Meijer, B. Moss, and E. Jeppesen. 1993. Alternative equilibria in shallow lakes. Trends in Ecology and Evolution 8: 275:279.

Simons, J., M. Ohm, R. Daalder, P. Boers, and W. Rip. 1994. Restoration of Botshol (The Netherlands) by reduction of external nutrient load: recovery of a characean community, dominated by Chara connivens. Hydrobiologia 275/276:243–253.

Stansfield, J., M. R. Perrow, L. D. Tench, A J. D. Jowitt and A. A. L. Taylor. 1997. Submerged macrophyrtes as refugees for grazing cladode against fish predation: observations on seasonal changes in relation to macrophyte cover and predation pressure. Hydrobiologia 342/343:229-240.

Timms, R.M. and B. Moss. 1984. Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in a shallow wetland ecosystem. Limnology and Oceanography 29:472-486.

United States Department of Agriculture, National Agricultural Statistics Service (USDA-NASS). 2004. 2002 Census of Agriculture: Volume 1: Geographic Area Series Part 15. Washington, DC USDA/NASS.

United States Environmental Protection Agency (USEPA). 1996. Environmental indicators of water quality in the United States. USEPA. Office of Water (4503F). U.S. Government Printing Office, Washington D.C., USA EPA 8410R-96-002.

van den Berg, M.S., A.W. Breukelaar, C. Breukers, H. Coops, R.W. Doef, and M.-L. Meijer. 1994. Vegetated areas with clear water in turbid shallow lakes. Aquatic Botany 49:193-196.

van den Berg, M.S., H. Coops, M.-L. Meijer, M. Scheffer, and J. Simons. 1998. Clear water associated with dense *Chara* vegetation in the shallow and turbid lake Veluwemeer, The Netherlands. Pages 339-352 *in* Jeppesen, E., Søndergaard, M., Christoffersen K. (editors), The structuring role of submerged macrophytes in lakes. Springer, New York.

van Donk, E., R.D. Gulati, and M.P. Grimm. 1989. Food-web manipulation in Lake Zwemlust: positive and negative effects during the first two years. Hydrobiological Bulletin 23:19-35.

van Donk, E. and W.J. van de Bund. 2002. Impact of submerged macrophytes including charophytes on phyto- and zooplankton communities: allelopathy versus other mechanisms. Aquatic Botany 72: 261-274.

Weaver, M.J., J.J. Magnuson, and M.K. Clayton. 1997. Distribution of littoral fishes in structurally complex macrophytes. Canada Journal of Fisheries and Aquatic Sciences 54: 2277-2289.

Weisner, S., G. Eriksson, W. Graneli, and L. Leonardson. 1994. Influence of macrophytes on nitrate removal in wetlands. Ambio 23:363-366.

Werner, E.E., J. F. Gilliam, D.J. Hall, and G.G. Mittelbach. 1983. An experimental test of the effects of predation risk on habitat use in fish. Ecology 64:1540-1548.

Werner, E.E. and D.J. Hall. 1988. Ontogenetic habitat shifts in bluegill: the foraging rate-predation rate trade-off. Ecology 69:1352-1366.

Wiley, M.J., R.W. Gorden, S.W. Waite, and T. Powless. 1984. The relationship between aquatic macrophytes and sport fish production in Illinois ponds: A simple model. North American Journal of Fisheries Management 4:111-119.

CHAPTER 2. ABIOTIC FACTORS INFLUENCE OF AQUATIC VEGETATION ABUNDANCE IN IOWA LAKES

A paper to be submitted to *Journal of Aquatic Plant Management*

MEGAN A. ERNST AND JOSEPH E. MORRIS

Department of Natural Resource Ecology and Management
Iowa State University
Ames, Iowa, USA 50011

ABSTRACT

In spite of the importance of aquatic vegetation to lakes, there are ongoing conflicts between the need to manage vegetation for multiple users of a lake and the need for aquatic vegetation for the aquatic biota. In 2007, a study was undertaken to assess the relationships between water quality and aquatic vegetation communities in 13 Iowa lakes. These lakes varied in location and fishery management protocols. The total number of emergent/floating aquatic vegetation species per lake varied from six to 14 species, while the total number of submerged aquatic vegetation species per lake varied from three to 11 species. Mean emergent/floating aquatic vegetation abundance and submerged aquatic vegetation were compared against physical-chemical parameters. There were four significant relationships between physical-chemical parameters (alkalinity, hardness, chlorophyll *a*, and temperature) and emergent/floating vegetation abundance and significant relationships between submerged aquatic vegetation and chlorophyll *a*, Secchi-depth, total suspended solids, and total Kjeldahl nitrogen. The lakes with the best values of physical-

chemical indicators typically had higher submerged aquatic vegetation abundance, but not necessarily diversity. The nMDS plot shows relationships the lakes have with emergent/ floating vegetation and submerged aquatic vegetation species as well as abundance. The emergent/floating aquatic vegetation ordination indicates that lakes Meadow, Greenfield, Anita, and Mormon Trail share similar plant species. The submerged aquatic vegetation nMDS plot reiterates the strong negative relationship between Secchi-depth and chlorophyll *a* levels, and lakes that share these characteristics. Overall, each lake seemingly similar at first, has many unique characteristics, making it difficult to set up a comprehensive guideline for all Iowa lakes vegetation management practices. By using simple linear regression, Shannon diversity index, and nMDS plots, managers can start to understand similarities and differences among lakes with reference to aquatic vegetation and physical-chemical parameters.

INTRODUCTION

The health of lake ecosystems is viewed differently based upon the how the system is used. Fisheries biologists focus on the management of the fisheries, trying to produce a balance between predator and prey, while limnologists focus on water quality and nutrient input. Aquatic vegetation helps maintain the overall integrity of aquatic ecosystems by playing a vital part in the autotrophic community and the cycling of nutrients (van Donk et al. 1989; Sand-Jensen and Borum 1991). Aquatic vegetation limits re-suspension of substrates and nutrients by reducing wind-driven sediments and discharge into shallow lake systems (Barko and James 1998) while also lowering nutrient concentrations (van Donk et al. 1989), increasing water clarity, producing oxygen, reducing shore erosion, and providing food and habitat for aquatic fauna (Canfield et al. 1984; Timms and Moss 1984; Jeppesen et al. 1990; Scheffer et al. 1993; Meijer et al. 1994; Moss et al. 1994; Egertson et al. 2004).

Aquatic vegetation abundance and distribution is influenced by environmental factors such as irradiance (Secchi-Depth), temperature, wave action, lake size, catchment basin morphology, and water chemistry (Gasith and Hoyer 1998). Traditionally aquatic vegetation abundance and distribution have been described using depth gradients in lakes. At the shallow depths vegetation growth is limited because of poor sediment and damaging wave action while at the deep end of the gradient, light is the limiting variable (Chambers and Kalff 1985; Duarte et al. 1986; Scheffer et al. 1992; Scheffer 2004). Middelboe and Markager (1997) research suggests that aquatic vegetation require different intensities of light (i.e., rosette-type angiosperms require the most light whereas charophytes require the least light).

Studies have presented positive relationships between water clarity and maximum depth of aquatic vegetation (Canfield et al. 1985; Chambers and Kalff 1985; Scheffer 2004). Scheffer (1990) theorized that in shallow lakes of similar depth, submerged aquatic vegetation will disappear at a critical turbidity threshold. Lougheed et al. (1998) concluded that at a threshold of 20 NTU, submerged aquatic vegetation was reduced to less than five species, whereas a more diverse vegetated community existed in clearer water. In the presence of common carp *Cyprinus carpio* and other benthivorous fish, aquatic vegetation abundance decreased while algal biomass and turbidity levels increased (Meijer et al 1990; Richardson et al. 1990; Breukelaar et al. 1994).

Water clarity is not affected by increased nutrient loading until a critical threshold is passed. Two indicators a lake is changing from a clear-water state with abundant submerged aquatic vegetation towards an enriched turbid-water stable state with free-floating and emergent vegetation are the sudden loss of transparency and reduction in species diversity (Scheffer 1990; Scheffer et al. 2002; Egertson et al. 2004).

Hypolimnetic anoxia allows phosphorus to recycle from the sediments to the epilimnion, upholding a turbid state. Phosphorus released from sediments is dependent on resuspension and biotrubation as well as the characteristic of the sediment, such as iron, aluminum, organic and total phosphorus (TP) concentrations (Phillips et al. 1994). Once the system has switched to a turbid-stable state, nutrient reduction alone may have little effect on clarity and plant regeneration. Instead food-web manipulation (e.g., fish biomass reduction) may be needed to return the lake to its clear-water stable state. Fish biomass reductions lessen sediment re-suspension and allow large-bodied, algae-eating zooplankton populations to increase (Scheffer et al. 1993).

Nutrient availability is reduced when aquatic vegetation is present. After drastic aquatic vegetation removal in Lake Conroe, Texas, algal biomass increased rapidly, cyanobacteria dominated summer blooms reducing water clarity, and major zooplankton taxa (cladoceran) biomass and bluegill *Lepomis macrochirus* biomass decreased (Bettoli et al 1991; Maceina et al. 1992). Barko et al. (1988) grew *Hydrilla* *verticillata* over two 6-week periods with results showing nutrient reduction by aquatic vegetation was greater than 90% of exchangeable nitrogen and reduced acid-extractable phosphorus from sediments by more than 30%. Rooney et al. (2003) found bulk sediment per unit area in vegetation beds to be twice that of the profundal zone, even though phosphorus levels were only 1/6th of that in the profundal zone, losing >70% sedimented phosphorus after deposition, creating a possible management tool to filter and protect water supplies from pollution.

In addition to altering the abiotic relationships in a lake, aquatic vegetation influences the biotic communities. For instance, a patchy distribution of aquatic vegetation is important in structuring a community that provides a variety of microhabitats supporting diverse fauna by limiting predator efficiency (Crowder and Cooper 1982; Weaver et al. 1997). Durocher et al. (1984) found that any reduction below 20% vegetation cover resulted in both a reduction in recruitment and standing crop of largemouth bass *Micropterus salmoides* in Texas reservoirs whereas ponds with nearly 50% vegetation cover of total lake area had high densities of age-0 largemouth bass which suggests that cover is a primary factor in bass survival.

In recent decades, pollutant inputs from agricultural and urban landscapes have increased and degraded water quality of rivers, lakes, and coastal oceans (Carpenter et al. 1998). Eutrophication, stemming from excessive nutrient loading of phosphorus and nitrogen, is the most common impairment of surface waters in the United States (U.S. EPA 1996), accounting for ~50% of all impaired lakes. In a region where there have been substantial changes in the natural landscape (e.g., Iowa) these changes have become more problematic.

Many of Iowa's aquatic systems are inundated with algae and aquatic macrophytes. Rich agricultural terrain (ca. 9.7 million hectares of tilled cropland; USDA 2004) contributes excess nutrients and soil loss from mismanaged watersheds aiding in the eutrophication of lakes and altering aquatic vegetation species and abundance (Scheffer et al. 2002; Egertson et al. 2004). Lakes with low nutrient content are often dominated by relatively small plants; lakes with elevated nutrient levels have a high abundance of aquatic vegetation that extends throughout the entire water column (Scheffer 2004).

This study's goal is to provide Iowa's lake managers the tools to quantify and evaluate the role of aquatic vegetation in Iowa's impoundments. These tools are needed as part of a comprehensive management protocol that managers can use to better manage Iowa's lakes for all user groups. The study objective was to assess aquatic vegetation abundance and diversity as they are related to physical-chemical parameters in Iowa's lakes.

STUDY AREA

Aquatic vegetation and environment data were collected monthly May to September 2007, from 13 Iowa lakes (impoundments). The thirteen study lakes (Lake Ahquabi, Lake Anita, Greenfield Lake, Lake Hendricks, Meadow Lake, Mormon Trail Lake, Pleasant Creek, Red Haw Lake, Silver Lake, Lake Smith, Swan Lake, Lake of Three Fires, and Lake Wapello) varied in size, depth, and grass carp abundance as well as presence of invasive plants species (e.g., curly leaf *Potamogeton crispus*) Lake Ahquabi, Greenfield Lake, Lake Hendricks, and Mormon

Trail Lake, and brittle naiad *Najas minor* in Meadow Lake and Pleasant Creek. Both invasive species were present in Lake Wapello (Table 1).

METHODS

Water Quality Collections- Water quality samples were collected bi-monthly May to September 2007 in limnetic waters near the dam structure. During the months of August and September 2007, additional samples located in littoral waters were collected. Sampling points were stationary, chosen randomly and located using a Garmin GPSmap 76CSX Global Positioning System (GPS). Collection was accomplished using an integrated 2-m tube sampler with a one-way check valve. Water quality analysis was completed by University of Iowa Hygienics Laboratory to assess levels of TP, total Kjeldahl nitrogen (TKN), total suspended solids (TSS), chlorophyll *a*, hardness, and alkalinity.

Aquatic Vegetation Collections- Aquatic vegetation surveys were conducted the first week of each month from May to September 2007. Stationary transect lines were randomly selected around the perimeter of each lake. Lakes $\leq$40.47 ha had 13 transects, lakes 40.48 ha -101.17 ha had 19 transects, and lakes 101.18 ha -202.34 ha had 25 transects (Quist et al. 2007).

Aquatic vegetation was categorized as either submerged (SAV) or emergent/floating (EFAV) along each transect line. The sampling device for SAV consisted of two welded garden rake heads measuring 35.6 cm in length and having 14, 5.1-cm teeth attached to an extendable 5.5-m push pole (Yin et al. 2000). It was lowered to the substrate, turned 180 degrees, raised, and pulled horizontally through the surface water to rinse and compact aquatic vegetation on the rake head. Total percent coverage was estimated using marked gradations on the teeth; percent species coverage was visually estimated for each sample. Emergent/floating aquatic vegetation was sampled by placing a floating, 1-m diameter hoop on the surface water and overall and species percent coverage were quantified for each sample. The percent coverage of the total sample of a particular species was calculated by multiplying overall percentage by that species coverage and dividing the product by 100. For instance, in a SAV sample that has 50% coverage and two species of plants that comprise of 20 and 80%, the subsequent individual coverage of the total sample is 10 and 40%, respectively.

Transects were sampled perpendicular from the waters edge outward at 0.61-m contour depth increments to a minimum of 2.4-m for both SAV and EFAV samples. Transects were complete when two consecutive rake samples were void of SAV past the 2.4-m mark, when depth contours indicated a decrease in water depth, or when depths reached 4.9-m. If aquatic vegetation was quantifiable at 4.9-m, one last pull occurred at 5.5-m and aquatic vegetation was noted as being present or absent.

Statistical Analysis- Mean overall SAV and EFAV abundances and abiotic parameters were calculated for each lake for 2007. The vegetation data (SAV and EFAV) were arcsine square root transformed prior to statistical analysis for heterogeneity of variance.

Simple linear regressions between transformed SAV abundance and mean environmental parameters (water temperature, pH, Secchi depth, TP, TKN, TKN:TP ratio, TSS, and chlorophyll *a*) to determine relationships were computed using JMP® 7.0.2, a statistical software package of SAS Institute (2007). Linear regressions were also performed between transformed EFAV and the same environmental parameters to determine any relationships. A probability level of 5% was used to determine statistical significance.

The Shannon index (H') measurement (Shannon 1948) was used to calculated aquatic vegetation diversity in our study lakes.

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

Ordination statistics were run using R-program version 2.5.1. The ordination technique used was non-metric multidimensional scaling (nMDS), with the Bray-Curtis distance equation to find relationships among lakes, SAV species, EFAV species, and water chemistry variables. Bray-Curtis ordination is used to determine site similarities based on samples from communities species composition and abundance (Bray and Curtis 1957). A nMDS stress level <10 indicates a strong ordination plot.

RESULTS

Mean water chemistry values collected from May 2007 to September 2007 are recorded in Table 2. The mean surface temperature varied between 22 ± 0.5 ^{o}C and 25 ± 0.5 ^{o}C for the thirteen lakes (Table 2). Lakes located in the northern latitudes (Lake Smith and Lake Hendricks) had cooler mean water temperatures (ca. 3^{o}C) than lakes located in the southern part of the state (Lake of Three Fires and

Lake Wapello). Mean pH levels varied from 7.5 ± 0.4 (Smith) to 9.2 ± 0.2 (Silver), while mean Secchi-depth varied from low light penetration of 40 ± 5 cm (Silver) to high light penetration of 162 ± 27 cm (Pleasant Creek; Table 2). Mormon Trail Lake, Pleasant Creek, and Red Haw Lake generally had the lowest mean TP levels of 0.02 ± <.01 mg/L and Swan Lake had the highest mean TP level of 0.16 ± 0.04 mg/L. Also, Red Haw Lake had the lowest mean TKN level of 0.88 mg/L and Silver Lake mean TKN level was the highest at 4.39 mg/L (Table 2). The ratio of TKN:TP in our study lakes varied from 15.5 to 59.4 with low ratios possible indicting a suitable environment for the presence of cyanobacteria (Table 2).

Two possible water chemistry variables that influence light penetration are TSS and chlorophyll *a*. Mean total suspended solid levels varied from 5 ± 1 mg/L (Red Haw) to 40 ± 14 mg/L (Swan), while mean chlorophyll *a* levels varied between 11 ± 2 μg/L (Red Haw) and 133 ± 26 μg/L (Smith; Table 2). Alkalinity varied from 75 ± 8 mg/L as $CaCO_3$ (Hendricks) to 147 ± 8 mg/L as $CaCO_3$ (Swan), while hardness varied from 76 ± 5 mg/L as $CaCO_3$ (Three Fires) to 195 ± 17 mg/L as $CaCO_3$ (Smith; Table 2).

The total number of EFAV species per lake varied from six to 14 species, while mean EFAV varied between 1 ± 0.1% (Smith) and 20 ± 0.4% (Wapello; Table 5). Emergent/floating aquatic vegetation abundance was the highest in Lake of Three Fires but the Shannon diversity index was only 5.4 (Table 5). The most diverse lake was Swan Lake with a Shannon index of 9.19, while Meadow Lake had the lowest Shannon index diversity of 3.11 (Table 5).

Mean EFAV abundance was compared against physical-chemical parameters in Tables 1 and 2. There were four significant relationships between water chemistry parameters (alkalinity, hardness, chlorophyll *a*, and temperature) and EFAV abundance. Alkalinity (r^2=0.37, P=0.03), hardness (r^2=0.50, P=0.01), and chlorophyll *a* (r^2=0.39, P=0.02) were negatively correlated with EFAV abundance, while temperature and EFAV abundance were positively correlated (r^2=0.32, P=0.05; Table 3, Figure 1).

The total number of SAV species per lake varied from three to 11 species, while mean SAV abundance varied from 0.3 ± <0.1% (Meadow) to 15 ± 0.6% (Red Haw; Table 6). In Lake Red Haw, coontail was present 67% of the samples and comprised 89% of the total SAV abundance. Even though SAV abundance was the highest in Red Haw Lake, its Shannon diversity index of 1.5 was the lowest of all lakes (Table 6). The most diverse lake was Mormon Trail Lake with a Shannon index of 6.95 (Table 6). Lake Ahquabi had an abundance of invasive species curly pondweed, present 24% of the samples and accounting for 64% of overall SAV abundance (Table 6).

Mean submerged aquatic vegetation abundance was also compared to physical-chemical parameters. There were four significant relationships between physical-chemical parameters (chlorophyll *a,* TKN, TSS, and Secchi-depth) (Table 3). Chlorophyll *a* (r^2 =0.47, P=0.01), TKN (r^2 =0.37, P=0.03), and TSS (r^2 =0.38, P=0.03) were negatively correlated to SAV abundance while Secchi-depth (r^2 =0.58, P=<0.01) was positively correlated (Table 4, Figure 2). In addition, total phosphorous and chlorophyll *a* levels had a significant positive relationship (adjusted r^2= 0.45, P=0.01).

The ordination technique nMDS, method Bray-Curtis (three dimensions, stress=8.00), was used to show interactions between the lakes and vegetation species, and the influences of the physical-chemical parameters on the lake communities (Figures 3 and 4.) Physical-chemical variables are represented as vectors. The direction of the vectors shows the path of the gradient while the length of the arrow shows correlation strength between the variable and the ordination (Table 7). The emergent/floating vegetation ordination indicates that the following pairs have similarities in EFAV species and abundance: Greenfield and Anita, Wapello and Hendricks, and Ahquabi and Red Haw have (Figure 3). By displaying lakes far apart, the ordination indicates that Three Fires and Smith Lake are very dissimilar in EFAV species and abundance (Figure 3). The following associations between lakes and vegetation species exists: Silver Lake and softstem bulrush *Schoenoplectus tabernaemontani* (SCTA2), Lake of Three Fires and prairie cordgrass *Spartina pectinata* (SPPE), Lake Greenfield and reed canarygrass *Phalaris arundinacea* (PHAR3), Lake Anita and arrowhead *Sagittaria* spp. (SAGIT), Swan Lake and American water-willow *Justicia americana* (JUAM), hardstem bulrush *Schoenoplectus acutus* (SCAC3), Lake Wapello and common spikerush *Eleocharis palustris* (ELPA3) and Lake Ahquabi and giant duckweed *Spirodela polyrrhiza* (SPPO), pondweed *Potamogeton* spp. (POTAM), water plantain *Alisma* spp (ALISM). Two physical-chemical parameter vectors (pH and TKN) have significant P-values (Table 7).

The nMDS plot (three dimensions, stress=8.54) of SAV shows a close relationship between Pleasant Creek and vegetation species American eelgrass *Vallisneria americana* (VAAM3) and brittle naiad (NAMI). Swan Lake has a close relationship with leafy pondweed *Potamogeton foliosus* (POFO3), while quillwort *Isoetes* spp. (ISOET) is closely affiliated with Mormon Trail Lake. Figure 4 also shows a strong relationship between curly pondweed (POCR3) and Lake Ahquabi. Some of the commonly observed species (e.g., coontail) are seen in the ordination plot as more centrally located among several lakes (Figure 4). Based on the length and direction of the physical-chemical vectors, Secchi-depth has a positive strong significant relationship with SAV abundance and species (P=0.03; Table 7). Secchi-depth is also negatively correlated with chlorophyll a levels, which is associated with lakes having little SAV such as Lake Smith, Lake of Three Fires, Meadow Lake and Silver Lake (Figure 4).

DISCUSSION

This study revealed several important results that can explain the inter-relationships between physical-chemical parameters and aquatic vegetation species and abundance. These relations can then be used by agency biologists to better manage the aquatic biota in Iowa lakes.

Some lakes had similar mean physical-chemical parameters (TSS levels in Lake Ahquabi and Greenfield Lake) while other lakes had drastic differences, (TKN levels in Red Haw Lake and Silver Lake). On average, Red Haw Lake, Pleasant Creek, Swan Lake, Lake Smith, and Silver Lake had the highest and lowest physical-chemical values. These characterizations exemplify that while all of our lakes are located in Iowa, there are still differences in their physical-chemical characteristics.

Emergent/floating vegetation abundance is influenced by water temperature, chlorophyll *a*, alkalinity, and hardness (Table 3 and Figure 1). A possible follow up study could be completed on why alkalinity and hardness affect EFAV, but one possibility is due to pH levels and buffering capacity. Another possibility is that this relationship might not be a causative effect as the two lakes with the highest EFAV populations, Lake of Three Fires and Wapello, has the lower alkalinity levels and their removal would have made linear relationship less significant. Temperature may be a seasonal and geographical location effect. Lake Wapello and Lake of Three Fires, the most southern of the study lakes, on average had the highest water temperature and the highest abundance of emergent/floating vegetation (Tables 2 and 5).

When turbidity is low, lakes tend to have more SAV present (Canfield et al. 1985; Chambers and Kalff 1985; Scheffer, 2004). Scheffer (2004) explains in the absence of aquatic vegetation, chlorophyll *a* increases with an increase of TP levels and our study lakes followed this trend (r^2= 0.45, P=0.01). This increase of chlorophyll and TP levels is likely to increase lake turbidity and phytoplankton abundance while hindering aquatic macrophyte growth. Water clarity indicators (e.g., low chlorophyll *a,* low TSS, and high Secchi-depth) all have a significant relationship with SAV abundance (Table 4 and Figure 1). The lakes with the best values of these indicators typically had higher SAV abundance, but not necessarily diversity (Tables

2 and 6). Lakes with high levels of chlorophyll *a*, TSS, and low Secchi-depth often had low amounts of SAV abundance (Table 2 and 6). Lake managers may use these three indicators as well as TKN as guidelines for SAV growing success. Even though none of my study lakes had vegetation abundance at the desired 20% (Durocher et al. 1984), some physical-chemical guidelines can be inferred from my research. Light reaching the substrate is important in the initial growth of aquatic vegetation and non-canopy forming plants (e.g., charophytes). Turbid, shallow waters are dominated with canopy forming species (e.g., sago pondweed *Potamogeton pectinatus*) because of their ability to grow near the surface (Barko and Smart 1981; Tanner et al. 1993) and store enough energy in their tubers and rhizomes during over-wintering to support early growth in spring during low light levels (Hodgson 1966). These studies may help explain why many of my study lakes have low abundance of charophytes (*Chara* spp.) and a higher abundance of pondweeds, (*Potamogeton crispus*, and *Potamogeton nodosus*) which grow near the water surface (Table 6). Additionally, similar to Bachmann et al. (2002), there was no predictable relationship in EFAV or SAV abundance and TP levels (Table 3 and 4). This may be due to the fact that aquatic vegetation obtains nutrients from sediments rather than from the water. However, TP should not be eliminated as a possible indicator of water quality, due to its intricate relationship with TKN and phytoplankton blooms.

The nMDS plot shows relationships the lakes have with emergent/ floating vegetation and submerged aquatic vegetation species as well as abundance (Figure 3 and 4). The EFAV ordination indicates that lakes Meadow, Greenfield, Anita, and

Mormon Trail share similar plant species; therefore, are plotted in the same quadrate. Interestingly, these four lakes are geographically close and are located in bordering counties of Adair and Cass. Lake biologists could manage lakes by using ordination techniques to group similar lakes together.

While Lake of Three Fires and Smith Lake are dissimilar in emergent/ floating vegetation abundance and species, they are more similar in submerged vegetation (Figures 3 and 4). Their similarities in the submerged vegetation nMDS plot is due to the fact both lakes have very little submerged vegetation (Table 4). Also the SAV nMDS plot reiterates the strong negative relationship between Secchi-depth and chlorophyll *a* levels, and lakes that share these characteristics (Figure 4). The right side of the X-axis on the nMDS plot shows lakes with little vegetation, high chlorophyll *a* levels, and low Secchi-depth (Figure 4). Lakes located in the left side of the X-axis on the nMDS plot often have more SAV abundance and speciation, deeper light penetration (Secchi-depth), and lower chlorophyll *a* levels. Lake Red Haw is positioned slightly away from other lakes probably due to its high vegetation abundance, but low species diversity (Figure 4.)

Overall, each lake seemingly similar at first, has many unique characteristics, making it difficult to set up a comprehensive guideline for all Iowa lakes vegetation management practices; but, by using simple linear regression, Shannon diversity index, and nMDS plots, managers can start to understand similarities and differences among lakes with reference to aquatic vegetation and physical-chemical parameters.

Management Implications

Managing aquatic vegetation in public lakes, agency staff often seek insight into physical-chemical parameters that infer possible management implications. In our study, lakes with chlorophyll *a* levels around 60 µg/L,TKN levels around 2 mg/L, Secchi-depth near 100 cm and TSS around 10 mg/L appear to be the limit between higher SAV abundance and lower SAV abundance. However, while our study did include 13 lakes of various physical and management scenarios, this suggested guideline must be approached with caution as other lakes might have different characteristics. In addition, since all of our lakes are impoundments, lakes with natural origins might have different management guidelines.

REFERENCES

Bachmannn, R.W., C.A. Horsburgh, M.V. Hoyer, L.K. Mataraza, and D.E. Canfield Jr. 2002. Relations between trophic state indicators and plant biomass in Florida lakes. Hydrobiologia 470: 219-234.

Barko, J. W. and W. F. James. 1998. Effects of submerged aquatic macrophytes on nutrient dynamics, sedimentation and resuspension. Pages 197-217 *in* Jeppesen E., Søndergaard M., Christoffersen K (Editors). The structuring role of submerged Macrophytes in Lakes. Springer, New York.

Barko, J.W. and R.M. Smart. 1981. Comparative influences of light and temperature on the growth and metabolism of selected submersed freshwater macrophytes. Ecological Monographs 51:219-236.

Barko, J.W., R.M. Smart, D.G. McFarland, and R.L. Chen. 1988. Interrelationships between the growth of *Hydrilla verticillata* (L.f.) Royle and sediment nutrient availability. Aquatic Botany 32: 205-216.

Bettoli, P.W., J.E. Morris, and R.L. Noble. 1991. Changes in the abundance of two atherinid species after aquatic vegetation removal. Transactions of the American Fisheries Society 120: 90-97.

Bray, J.R. and J.T. Curtis. 1957. An ordination of upland forest communities of southern Wisconsin. Ecological Monographs 27: 326-349.

Breukelaar, A.W., E.H.R.R. Lammens, J.G.P.K. Breteler, and I. Tatrai. 1994. Effects of benthivorous bream (*Abramis brama*) and carp (*Cyprinus carpio*) on sediment resuspension and concentrations of nutrients and chlorophyll *a*. Freshwater Biology 32:113-121.

Canfield, D.E. Jr., K.A. Langeland, S.B. Linda, and W.T. Haller. 1985. Relations between water transparency and maximum depth of macrophyte colonization in lakes. Journal of Aquatic Plant Management 23:25-28.

Canfield, D. E., J.V. Shireman, D.E. Colle, W. T. Haller, C. E. Watkins, and M. J. Maceina. 1984. Prediction of chlorophyll *a* concentrations in Florida lakes: importance of aquatic macrophytes. Canadian Journal of Fisheries and Aquatic Sciences 41:497-501.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8: 559-568.

Chambers, P.A. and J. Kalff. 1985. Depth distribution and biomass of submersed aquatic macrophyte communities in relation to secchi depth. Canadian Journal of Fisheries and Aquatic Sciences 42:701–709.

Crowder, L.B. and W.E. Cooper. 1982. Habitat structural complexity and the interaction between bluegills and their prey. Ecology 63:1802-1813.

Duarte, C.M., J. Kalff, and R.H. Peters.1986. Patterns in biomass and cover of lake macrophytes. Canadian Journal of Fisheries and Aquatic Sciences 43:1900–1908.

Durocher, P.P., W.C. Provine, and J.E. Kraii. 1984. Relationship between abundance of largemouth bass and submerged vegetation in Texas reservoirs. North American Journal of Fisheries Management 4:84-88.

Egertson, C. J., J.A. Kopaska, and J.A. Downing. 2004. A century of change in macrophyte abundance and composition in response to agricultural eutrophication. Hydrobiologia 524:145-156.

Gasith, A. and M.V. Hoyer. 1998. Structuring role of macrophytes in lakes: changing influence along lake size and depth gradients. Pages 381-389 *in* Jeppesen, E., Søndergaard, M., Christoffersen K. (editors), The structuring role of submerged macrophytes in lakes. Springer, New York.

Hodgson, R.H. 1966. Growth and carbohydrate status of sago pondweed. Weeds 14:263-268.

Jeppesen, E., J.P. Jensen, P. Kristensen, M. Søndergaard, E. Mortensen, O. Sortkjaer, and K. Olrik. 1990. Fish manipulation as a lake restoration tool in shallow, eutrophic, temperate lakes 2: Threshold levels, long-term stability and conclusions. Hydrobiologia 200/201:219-227.

Lougheed, V.L., B. Crosbie, and P. Chow-Fraser. 1998. Predictions on the effect of common carp (*Cyprinus carpio*) exclusion on water quality, zooplankton and submergent macrophytes in a Great Lakes wetland. Canadian Journal of Fisheries and Aquatic Sciences 55:1189-1197.

Maceina, M.J., M.F. Cichra, R.K. Betsill, and P.W. Bettoli. 1992. Limnological changes in a large reservoir following vegetation removal by grass carp. Journal of Freshwater Ecology 7:81-95.

Meijer, M-L., M.W. de Haan, A.W. Breukelaar, and H. Buiteveld. 1990. Is reduction of the benthivorous fish and important cause of high transparency following biomanipulation in shallow lakes? Hydrobiologia 200/201:303-315.

Meijer, M-L., E. Jeppesen, E. van Donk, B. Moss, M. Scheffer, E. Lammens, E. van Nes, J.A. van Berkum, G.J. de Jong, B.A. Faafeng, and J.P. Jensen. 1994. Long-term responses to fish-stock reduction in small shallow lakes: interpretation of five-year results of four biomanipulation cases in The Netherlands and Denmark. Hydrobiologia 275/276:457-466.

Middelboe, A.L. and S. Markager. 1997. Depth limits and minimum light requirements of freshwater macrophytes. Freshwater Biology 37: 553-568.

Moss, B., S. McGowan, and L. Carvalho. 1994. Determination of phytoplankton crops by top-down and bottom-up mechanisms in a group of English lakes, the West Midland Meres. Limnology and Oceanography 39:1020-1029.

Phillips, G., R. Jackson, C. Bennett, and A. Chilvers. 1994. The importance of sediment phosphorus release in the restoration of very shallow lakes (The Norfolk Broads, England) and the implications for biomanipulation. Hydrobiologia 275/276:445-456.

Quist, M.C., L. Bruce, K. Bogenschutz, and J.E. Morris. 2007. Sample size requirements for estimating species richness of aquatic vegetation in Iowa lakes. Journal of Freshwater Ecology 22:477-492.

Richardson, W.B., S.A. Wickham, and S.T. Threlkeld. 1990. Food-web response to the experimental manipulation of a benthivore (*Cyprinus carpio*), zooplanktivore (*Menidia beryllina*) and benthic insects. Archiv für Hydrobiologie 119:143-165.

Rooney, N., J. Kalff, and C. Habel. 2003. The role of submerged macrophyte beds in phosphorus and sediment accumulation in Lake Memphremagog, Quebec, Canada. Limnology and Oceanography 48:1927-1937.

Sand-Jensen, K. and J. Borum. 1991. Interactions among phytoplankton, periphyton, macrophytes in temperate freshwaters and estuaries. Aquatic Botany 41: 137-151.

Scheffer, M. 1990. Multiplicity of stable states in freshwater ecosystems. Hydrobiologia 200/201:475-486.

Scheffer, M. 2004. Ecology of shallow lakes. Chapman and Hall, London, pp 225-357.

Scheffer, M., S. Carpenter, J.A. Foley, C. Folke and B. Walker. 2002. Catastrophic shifts in ecosystems. Nature 413: 591-596.

Scheffer, M., S.H. Hosper, M.L. Meijer, B. Moss, and E. Jeppesen, 1993. Alternative equilibria in shallow lakes. Trends in ecology and evolution(TREE) 8:275-279.

Scheffer, M., M.R. De Redelijkheid, and F. Noppert. 1992. Distribution and dynamics of submerged vegetation in a chain of shallow eutrophic lakes. Aquatic Botany 42:199-216.

Shannon, C.E. 1948. A mathematical theory of communication. Bell System Technical Journal 27: 379-423, 623-656.

Tanner, C.C., J.S. Clayton, and R.D.S. Wells. 1993. Effects of suspended solids on the establishment and growth of *Egeria densa*. Aquatic Biology 45:299-310.

Timms R. M. and B. Moss. 1984. Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in shallow wetland ecosystem. Limnology and Oceanography 29:472-486.

United States Department of Agriculture, National Agricultural Statistics Service (USDA-NASS). 2004. 2002 Census of Agriculture: Volume 1: Geographic Area Series Part 15. Washington, DC USDA/NASS.

United States Environmental Protection Agency (USEPA). 1996. Environmental indicators of water quality in the United States. USEPA. Office of Water(4503F). U.S. Government Printing Office, Washington D.C., USA EPA 8410R-96-002.

van Donk, E., R. D. Gulati, and M.P. Grimm. 1989. Food-web manipulation in Lake Zwemlust: positive and negative effects during the first two years. Hydrobiology Bulletin 23:19-35.

Weaver, M.J., J.J. Magnuson, and M.K. Clayton. 1997. Distribution of littoral fishes in structurally complex macrophytes. Canada Journal of Fisheries and Aquatic Sciences 54: 2277-2289.

Yin, Y., J.S. Winkelman and H.A. Langrehr. 2000. Long Term Resource Monitoring Program procedures: Aquatic vegetation monitoring. U.S. Geological Survey, Upper Midwest Environmental Sciences Center, La Crosse, WI, USA, LTRMP 95-P002-7.

Table 1. Summary information for the 13 Iowa study lakes in 2007. Information summarized is: Lake, county, mean depth (m), lake size (ha), density of grass carp *Ctenopharyngodon idella* (fish/ha).

Lake	**County**	**Mean Depth (m)**	**Lake size (ha)**	**Grass Carp density (fish/ha)**
Lake Ahquabi	Warren	2.99	47.29	None
Lake Anita	Cass	3.77	70.9	None
Greenfield Lake	Adair	3.08	19.59	9.1
Lake Hendricks	Howard	2.35	19.43	1.1
Meadow Lake	Adair	3.11	14	14.2
Mormon Trail Lake	Adair	4.21	13.06	3.8
Pleasant Creek	Linn	4.96	163.67	8.5
Red Haw Lake	Lucas	4.44	30.63	None
Silver Lake	Delaware	1.95	15.96	None
Lake Smith	Kossuth	1.66	22.91	0.65
Swan Lake	Carroll	1.3	40.47	1.2
Lake of Three Fires	Taylor	2.52	39.08	None
Lake Wapello	Davis	3.94	114.33	1.6

Table 2. Summary statistics for environmental parameters measured during the sampling of emergent/floating and submerged aquatic vegetation in 13 Iowa lakes from May 2007 to September 2007.

Lake	Water Temperature (°C) ± SE	pH ± SE	Secchi Depth (cm) ± SE	Total Phosphorus (mg/L) (TP) ± SE	Total Kjeldahl Nitrogen (mg/L) (TKN) ± SE	TKN:TP	Total Suspended Solids (mg/L) ± SE	Chlorophyll *a* (µg/L) ± SE	Alkalinity (mg/L as $CaCO_3$) ± SE	Hardness (mg/L as $CaCO_3$) ± SE
Lake Ahquabi	25 ± 0.3	8.4 ± 0.08	104 ± 24	0.03 ± <0.01	1.24 ± 0.19	48.7	8 ± 2	50 ± 17	88 ± 5	95 ± 7
Lake Anita	24 ± 0.5	8.6 ± 0.08	94 ± 21	0.03 ± <0.01	1.39 ± 0.12	50.7	10 ± 1	44 ± 12	112 ± 3	130 ± 4
Greenfield Lake	23 ± 0.3	8.5 ± 0.06	102 ± 17	0.03 ± <0.01	1.26 ± 0.10	40.2	9 ± 2	23 ± 8	113 ± 5	129 ± 5
Lake Hendricks	22 ± 0.5	9.0 ± 0.21	106 ± 19	0.03 ± 0.01	1.30 ± 0.14	40.1	8 ± 1	44 ± 11	75 ± 8	119 ± 14
Meadow Lake	24 ± 0.4	8.8 ± 0.07	46 ± 7	0.05 ± 0.01	1.91 ± 0.15	37.9	26 ± 3	76 ± 14	96 ± 5	145 ± 43
Mormon Trail Lake	23 ± 0.5	8.4 ± 0.05	108 ± 9	0.02 ± <0.01	1.01 ± 0.06	59.4	7 ± 1	21 ± 4	118 ± 2	123 ± 4
Pleasant Creek	21 ± 0.3	8.6 ± 0.03	162 ± 27	0.02 ± <0.01	0.92 ± 0.05	56.3	6 ± 1	15 ± 4	123 ± 3	134 ± 3
Red Haw Lake	24 ± 0.4	8.1 ± 0.05	145 ± 19	0.02 ± <0.01	0.89 ± 0.07	54.5	5 ± 1	11 ± 2	85 ± 3	90 ± 2
Silver Lake	23 ± 0.3	9.2 ± 0.16	40 ± 5	0.12 ± 0.01	4.39 ± 0.30	36.4	28 ± 6	115 ± 23	87 ± 1	93 ± 3
Lake Smith	22 ± 0.5	7.5 ± 0.40	41 ± 5	0.06 ± 0.01	3.19 ± 0.26	51.2	22 ± 2	133 ± 26	143 ± 11	195 ± 17
Swan Lake	23 ± 0.4	9.0 ± 0.08	47 ± 9	0.16 ± 0.04	2.52 ± 0.30	15.5	40 ± 14	82 ± 24	147 ± 8	162 ± 5
Lake of Three Fires	25 ± 0.5	9.0 ± 0.20	84 ± 24	0.09 ± 0.01	1.57 ± 0.16	18.2	19 ± 4	47 ± 12	76 ± 4	76 ± 5
Lake Wapello	25 ± 0.2	8.5 ± 0.05	102 ± 10	0.02 ± <0.01	1.12 ± 0.10	55.1	7 ± 1	27 ± 10	85 ± 2	99 ± 6

Table 3. Summary correlations (r^2) of simple linear regressions between arcsine(square root) transformed emergent/floating vegetation abundance and limnetic environmental parameters in 13 Iowa Lakes in 2007. Statistical significant relationships between emergent/floating vegetation abundance and physical-chemical parameters are determined by ($P \leq 0.05$). Significant values are in bold.

Physical-chemical parameters	r^2	**P-value**
Temperature (^oC)	0.32	**0.05**
pH	0.23	0.09
Secchi Depth (cm)	0.08	0.34
Total Phosphorus (mg/L)	0.04	0.53
Total Kjeldahl Nitrogen (mg/L)	0.15	0.18
TKN:TP	0.01	0.70
Total Suspend Solids (mg/L)	0.09	0.31
Chlorophyll *a* (µg/L)	0.33	**0.04**
Alkalinity (mg/L as CaCO3)	0.37	**0.03**
Hardness (mg/L as CaCO3)	0.50	**0.01**
Lake Size(ha)	0.03	0.54
Lake depth(m)	0.16	0.17

Table 4. Summary correlations (r^2) of simple linear regressions between arcsine(square root) transformed submerged vegetation abundance and physical-chemical parameters in 13 Iowa Lakes in 2007. Statistical significant relationships between submerged vegetation abundance and environmental parameters are determined by ($P \leq 0.05$). Significant values are in bold.

Physical-chemical parameters	r^2	**P-value**
Temperature (^oC)	0.01	0.79
pH	<.01	0.95
Secchi Depth (cm)	0.57	**<.01**
Total Phosphorus (mg/L)	0.18	0.15
Total Kjeldahl Nitrogen (mg/L)	0.35	**0.03**
TKN:TP	0.13	0.23
Total Suspend Solids (mg/L)	0.37	**0.03**
Chlorophyll *a* (µg/L)	0.47	**0.01**
Alkalinity (mg/L as CaCO3)	0.03	0.55
Hardness (mg/L as CaCO3)	0.10	0.30
Lake Size(ha)	0.12	0.24
Lake depth(m)	0.20	0.13

Table 5. Emergent/floating aquatic vegetation summary statistics for the 13 Iowa lakes during the months of May 2007 to September 2007. Information summarized is: number of samples, mean emergent/floating vegetation abundance and standard error (SE), Shannon Index of diversity, species present, number of times that species was identified (N), individual species abundance and SE, and the percent each species contributes to the overall emergent/floating vegetation abundance. Abundance index (%) <0.1= trace (tr).

Lake	Number of Samples	Mean Emergent/Floating Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Ahquabi	565	5 ± 0.3	3.74							
				-----	Filamentous algae	ALGA	48	9%	3.2 ± 0.58	66%
				Alisma spp.	Water plantain	ALISM	1	<1%	tr	<1%
				Lemna minor	Common duckweed	LEMI3	5	1%	0.2 ± 0.09	3%
				Potamogeton nodosus	Longleaf pondweed	PONO2	16	3%	0.4 ± 0.17	8%
				Potamogeton spp.	Pondweed	POTAM	3	1%	0.5 ± 0.31	11%
				Sagittaria	Arrowhead	SAGIT	1	<1%	0.1 ± 0.12	3%
				Spirodela polyrrhiza	Giant duckweed	SPPO	9	2%	0.4 ± 0.20	8%
				Wolffia Columbiana	Watermeal	WOCO	1	<1%	tr	1%

Table 5. (Continued)

Lake	Number of Samples	Mean Emergent/Floating Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Anita	529	13 ± 0.2	7.18							
				-----	Filamentous algae	ALGAE	108	20%	4.3 ± 0.62	32%
				Carex spp.	Sedge	CAREX	1	<1%	tr	<1%
				Eleocharis	Spikerush	ELEOC	8	2%	0.2 ± 0.09	1%
				Eleocharis palustris	Common spikerush	ELPA3	1	<1%	tr	<1%
				Equisetum fluviatile	water horsetail	EQFL	1	<1%	tr	<1%
				Lemna minor	Common duckweed	LEMI3	103	19%	1.1 ± 0.28	8%
				Leersia oryzoides	Rice cutgrass	LEOR	4	1%	tr	<1%
				Phalaris arundinacea	Reed canarygrass	PHAR3	128	24%	5.5 ± 0.66	41%
				Polygonum amphibium	Water knotweed	POAM8	3	1%	tr	<1%
				Polygonum spp.	knotweed	POLYG4	1	<1%	tr	<1%
				Potamogeton nodosus	Longleaf pondweed	PONO2	31	6%	0.7 ± 0.19	5%
				Sagittaria cuneata	Arumleaf arrowhead	SACU	1	<1%	tr	<1%
				Sagittaria spp.	Arrowhead	SAGIT	8	2%	0.1 ± 0.03	1%
				Schoenoplectus fluviatilis	River bulrush	SCFL11	4	1%	0.2 ± 0.19	2%
				Spirodela polyrrhiza	Giant duckweed	SPPO	71	13%	0.3 ± tr	2%
				Wolffia columbiana	Watermeal	WOCO	86	16%	0.9 ± 0.27	7%

Table 5. (Continued)

Lake	Number of Samples	Mean Emergent/Floating Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Greenfield	334	14 ± 0.4	5.36							
				-----	Filamentous algae	ALGAE	69	21%	4.4 ± 0.78	32%
				Carex spp.	Sedge	CAREX	3	1%	0.3 ± 0.19	2%
				Lemna minor	Common duckweed	LEMI3	26	8%	0.5 ± 0.17	3%
				Phalaris arundinacea	Reed canarygrass	PHAR3	71	21%	7.3 ± 1.06	52%
				Polygonum spp.	knotweed	POLYG4	1	<1%	tr	<1%
				Sagittaria Spp.	Arrowhead	SAGIT	3	1%	0.4 ± 0.26	3%
				Sagittaria latifolia	Broadleaf arrowhead	SALA2	1	<1%	0.1 ± 0.11	1%
				Schoenoplectus fluviatilis	River bulrush	SCFL11	1	<1%	tr	<1%
				Schoenoplectus tabernaemontani	Softstem bulrush	SCTA2	5	2%	0.1 ± 0.09	1%
				Spirodela polyrrhiza	Giant duckweed	SPPO	20	6%	0.4 ± 0.16	3%
				Typha spp.	Cattail	TYPHA	5	2%	0.4 ± 0.24	3%
				-----	Unknown	UNKWN	5	2%	0.2 ± 0.09	1%
Hendricks	396	7 ± 0.3	4.52							
				-----	Filamentous algae	ALGAE	36	9%	3.8 ± 0.76	56%
				Eleocharis spp.	Spikerush	ELEOC	1	<1%	tr	<1%
				Lemna minor	Common duckweed	LEMI3	61	15%	1.6 ± 0.44	25%
				Leersia oryzoides	Rice cutgrass	LEOR	2	1%	tr	<1%
				Phalaris arundinacea	Reed canarygrass	PHAR3	1	<1%	tr	<1%
				Potamogeton spp.	Pondweed	POTAM	1	<1%	0.1 ± 0.06	1%
				Potamogeton nodosus	Longleaf pondweed	PONO2	5	1%	0.1 ± 0.05	1%
				Sagittaria cuneata	Arumleaf arrowhead	SACU	2	1%	0.1 ± 0.04	1%
				Sagittaria latifolia	Broadleaf arrowhead	SALA2	7	2%	0.1 ± 0.06	2%
				Wolffia columbiana	Watermeal	WOCO	44	11%	0.9 ± 0.25	13%

Table 5. (Continued)

Lake	Number of Samples	Mean Emergent/Floating Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Meadow	247	9 ± 0.6	3.11							
				-----	Filamentous algae	ALGAE	20	8%	0.6 ± 0.30	6%
				Lemna minor	Common duckweed	LEMI3	3	1%	tr	<1%
				Phalaris arundinacea	Reed canarygrass	PHAR3	57	23%	7.9 ± 1.30	90%
				Polygonum amphibium	Water knotweed	POAM8	1	<1%	tr	<1%
				Potamogeton nodosus	Longleaf pondweed	PONO2	1	<1%	tr	<1%
				Sagittaria spp.	Arrowhead	SAGIT	16	6%	0.3 ± 0.14	4%
Mormon Trail	403	7 ± 0.3	6.81							
				-----	Filamentous algae	ALGAE	22	5%	0.1 ± 0.04	2%
				Carex spp.	Sedge	CAREX	2	1%	tr	<1%
				Equisetum fluviatile	water horsetail	EQFL	2	1%	tr	<1%
				Lemna minor	Common duckweed	LEMI3	22	5%	0.2 ± 0.06	3%
				Leersia oryzoides	Rice cutgrass	LEOR	2	1%	0.1 ± 0.08	1%
				Phalaris arundinacea	Reed canarygrass	PHAR3	61	15%	5.2 ± 0.87	77%
				Potamogeton nodosus	Longleaf pondweed	PONO2	18	4%	0.5 ± 0.19	7%
				Sagittaria cuneata	Arumleaf arrowhead	SACU	1	<1%	tr	<1%
				Schoenoplectus fluviatilis	River bulrush	SCFL11	4	1%	tr	1%
				Spirodela polyrrhiza	Giant duckweed	SPPO	7	2%	tr	1%
				Typha spp.	Cattail	TYPHA	33	8%	0.6 ± 0.14	9%
				Wolffia columbiana	Watermeal	WOCO	1	<1%	tr	<1%
				-----	Unknown	UNKWN	3	1%	tr	1%

Table 5. (Continued)

Lake	Number of Samples	Mean Emergent/Floating Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Pleasant Creek	612	6 ± 0.2	8.48							
				-----	Filamentous algae	ALGAE	26	4%	1.3 ± 0.30	15%
				Carex spp.	Sedge	CAREX	1	<1%	0.1 ± 0.07	1%
				Eleocharis spp.	Spikerush	ELEOC	5	1%	0.2 ± 0.11	2%
				Lemna minor	Common duckweed	LEMI3	31	5%	0.7 ± 0.20	8%
				Leersia oryzoides	Rice cutgrass	LEOR	1	<1%	0.1 ± 0.12	1%
				Phalaris arundinacea	Reed canarygrass	PHAR3	8	1%	0.2 ± 0.10	2%
				Polygonum spp.	Knotweed	POLYG4	1	<1%	tr	<1%
				Potamogeton nodosus	Longleaf pondweed	PONO2	47	8%	2.5 ± 0.44	29%
				Sagittaria cuneata	Arumleaf arrowhead	SACU	14	2%	0.1 ± 0.05	2%
				Sagittaria spp.	Arrowhead	SAGIT	12	2%	0.2 ± 0.07	2%
				Sagittaria latifolia	Broadleaf arrowhead	SALA2	21	3%	0.6 ± 0.16	7%
				Schoenoplectus tabernaemontani	Softstem bulrush	SCTA2	2	<1%	tr	<1%
				Typha latifolia	Broadleaf cattail	TYLA	1	<1%	tr	<1%
				Typha spp.	Cattail	TYPHA	3	<1%	0.3 ± 0.17	3%
				Wolffia columbiana	Watermeal	WOCO	4	1%	0.1 ± 0.04	1%
Red Haw	501	9 ± 0.3	4.97							
				-----	Filamentous algae	ALGAE	27	5%	2.0 ± 0.46	21%
				Lemna minor	Common duckweed	LEMI3	8	2%	0.2 ± 0.12	2%
				Potamogeton nodosus	Longleaf pondweed	PONO2	50	10%	1.8 ± 0.33	19%
				Sagittaria Spp.	Arrowhead	SAGIT	1	<1%	tr	<1%
				Spirodela polyrrhiz	Giant duckweed	SPPO	51	10%	2.0 ± 0.33	21%
				Typha spp.	Cattail	TYPHA	9	2%	0.5 ± 0.24	6%
				Wolffia columbiana	Watermeal	WOCO	60	12%	2.8 ± 0.44	30%

Table 5. (Continued)

Lake	Number of Samples	Mean Emergent/Floating Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Silver	245	8 ± 0.4	6.9							
				-----	Filamentous algae	ALGAE	4	2%	0.2 ± 0.13	2%
				Carex spp.	Sedge	CAREX	2	1%	tr	<1%
				Eleocharis spp.	Spikerush	ELEOC	25	10%	2.0 ± 0.56	24%
				Lemna minor	Common duckweed	LEMI3	5	2%	tr	<1%
				Phalaris arundinacea	Reed canarygrass	PHAR3	26	11%	2.9 ± 0.78	34%
				Polygonum spp.	Knotweed	POLYG4	3	1%	tr	<1%
				Schoenoplectus	Bulrush	SCHOE6	1	<1%	tr	<1%
				Schoenoplectus tabernaemontani	Softstem bulrush	SCTA2	20	8%	1.7 ± 0.56	21%
				Typha latifolia	Broadleaf cattail	TYLA	1	<1%	0.1 ± 0.10	1%
				Typha spp.	Cattail	TYPHA	6	2%	0.3 ± 0.12	3%
				------	Unknown	UNKWN	26	11%	1.2 ± 0.41	14%
Smith	310	1 ± 0.1	6.65							
				-----	Filamentous algae	ALGAE	5	2%	0.1 ± 0.04	6%
				Carex spp.	Sedge	CAREX	3	1%	tr	3%
				Lemna minor	Common duckweed	LEMI3	4	1%	tr	1%
				Phalaris arundinacea	Reed canarygrass	PHAR3	20	6%	0.5 ± 0.17	37%
				Polygonum amphibium	Water knotweed	POAM8	1	<1%	tr	<1%
				Potamogeton nodosus	Longleaf pondweed	PONO2	2	1%	tr	1%
				Sagittaria spp.	Arrowhead	SAGIT	1	<1%	tr	<1%
				Spirodela polyrrhiza	Giant duckweed	SPPO	10	3%	0.5 ± 0.26	37%
				Typha spp.	Cattail	TYPHA	10	3%	0.2 ± 0.13	14%
				-----	Unknown	UNKWN	2	1%	tr	<1%

Table 5. (Continued)

Lake	Number of Samples	Mean Emergent/Floating Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Swan	254	4 ± 0.2	9.19							
				-----	Filamentous algae	ALGAE	18	7%	1.3 ± 0.43	32%
				Carex spp.	Sedge	CAREX	5	2%	0.1 ± 0.04	2%
				Justicia americana	American water-willow	JUAM	9	4%	0.2 ± 0.09	3%
				Lemna minor	Common duckweed	LEMI3	11	4%	1.1 ± 0.53	25%
				Phalaris arundinacea	Reed canarygrass	PHAR3	14	6%	0.9 ± 0.27	21%
				Polygonum amphibium	Water knotweed	POAM8	1	<1%	tr	<1%
				Polygonum spp.	Knotweed	POLYG4	2	1%	tr	<1%
				Sagittaria cuneata	Arumleaf arrowhead	SACU	2	1%	tr	<1%
				Sagittaria latifolia	Broadleaf arrowhead	SALA2	5	2%	0.5 ± 0.37	11%
				Schoenoplectus acutus	Hardstem bulrush	SCAC3	1	<1%	0.1 ± 0.05	1%
				Schoenoplectus fluviatilis	River bulrush	SCFL11	1	<1%	tr	<1%
				Schoenoplectus	Bulrush	SCHOE6	3	1%	tr	1%
				Typha spp.	Cattail	TYPHA	1	<1%	tr	1%
				-----	Unknown	UNKWN	3	1%	0.1 ± 0.14	3%
Three Fires	154	21 ± 1.0	5.41							
				-----	Filamentous algae	ALGAE	24	16%	2.8 ± 0.91	13%
				Eleocharis spp.	Spikerush	ELEOC	1	1%	tr	<1%
				Lemna minor	Common duckweed	LEMI3	28	18%	3.5 ± 1.25	17%
				Nelumbo lutea	American lotus	NELU	21	14%	7.2 ± 1.88	35%
				Phalaris arundinacea	Reed canarygrass	PHAR3	24	16%	5.6 ± 1.47	27%
				Polygonum spp.	Knotweed	POLYG4	7	5%	0.7 ± 0.35	3%
				Sagittaria spp.	Arrowhead	SAGIT	1	1%	0.1 ± 0.13	1%
				Spartina pectinata	Prairie cordgrass	SPPE	1	1%	0.4 ± 0.42	2%
				-----	Unknown	UNKWN	1	1%	0.3 ± 0.26	1%

Table 5. (Continued)

Lake	Number of Samples	Mean Emergent/Floating Vegetation Abundance Index (%)			Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)			% Species Composition of overall abundance
Wapello	691	20	±	0.4	3.66									
						-----	Filamentous algae	ALGAE	33	5%	1.9	±	0.42	10%
						Eleocharis palustris	Common spikerush	ELPA3	1	<1%	0.1	±	0.07	<1%
						Lemna minor	Common duckweed	LEMI3	2	<1%		tr		<1%
						Nelumbo lutea	American lotus	NELU	259	37%	12.3	±	0.96	63%
						Potamogeton nodosus	Longleaf pondweed	PONO2	52	8%	1.8	±	0.36	9%
						Sagittaria spp.	Arrowhead	SAGIT	4	1%	0.1	±	0.03	<1%
						Spirodela polyrrhiza	Giant duckweed	SPPO	46	7%	2.0	±	0.40	10%
						Typha spp.	Cattail	TYPHA	1	<1%		tr		<1%
						Wolffia columbiana	Watermeal	WOCO	36	5%	1.5	±	0.32	8%

Table 6. Submerged aquatic vegetation summary statistics for the 13 Iowa lakes during the months of May 2007 to September 2007. Information summarized is: number of samples, mean submerged vegetation abundance and standard error (SE), Shannon Index of diversity, species present, number of times that species was identified (N), individual species abundance and SE, and the percent each species contributes to the overall submerged vegetation abundance. Abundance index (%) <0.1= trace (tr).

Lake	Number of Samples	Mean Submerged Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Ahquabi	568	7 ± 0.2	6.61							
				-----	Filamentous algae	ALGA	50	9%	0.7 ± 0.17	10%
				Ceratophyllym demersum	Coontail	CEDE4	7	1%	tr	1%
				Chara spp.	Muskgrass	CHARA	44	8%	0.5 ± 0.12	8%
				Elodea canadensis	Canadian waterweed	ELCA7	13	2%	0.1 ± 0.03	1%
				Heteranthera dubia	Water star-grass	HEDU2	25	4%	0.2 ± 0.06	2%
				Najas flexilis	Nodding waternymph	NAFL	9	2%	0.1 ± 0.03	1%
				Potamogeton crispus	Curly pondweed	POCR3	134	24%	4.4 ± 0.65	64%
				Potamogeton nodosus	Longleaf pondweed	PONO2	6	1%	0.1 ± 0.05	1%
				Potamogeton pusillus	Small pondweed	POPU7	20	4%	0.3 ± 0.10	4%
				Stuckenia pectinata	Sago pondweed	STPE15	19	3%	0.5 ± 0.21	7%
				-----	Unknown	UNKWN	5	1%	0.1 ± 0.04	1%

Table 6. (Continued)

Lake	Number of Samples	Mean Submerged Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Anita	527	3 ± 0.1	5.4							
				-----	Filamentous algae	ALGA	181	34%	1.2 ± 0.18	37%
				Ceratophyllym demersum	Coontail	CEDE4	93	18%	0.8 ± 0.19	23%
				Chara spp.	Muskgrass	CHARA	10	2%	tr	1%
				Najas guadalupensis	Southern waternymph	NAGU	75	14%	0.8 ± 0.25	23%
				Potamogeton nodosus	Longleaf pondweed	PONO2	42	8%	0.4 ± 0.11	11%
				Potamogeton spp.	Pondweed	POTAM	28	5%	0.1 ± 0.02	2%
				Stuckenia pectinata	Sago pondweed	STPE15	43	8%	0.1 ± 0.04	4%
				Zannichellia palustris	Horned pondweed	ZAPA	1	0%	tr	0%
				-----	Unknown	UNKWN	1	0%	tr	0%
Greenfield	334	5 ± 0.3	3.34							
				-----	Filamentous algae	ALGA	104	31%	1.0 ± 0.17	20%
				Ceratophyllym demersum	Coontail	CEDE4	20	6%	0.3 ± 0.20	5%
				Chara spp.	Muskgrass	CHARA	4	1%	tr	0%
				Najas guadalupensis	Southern waternymph	NAGU	151	45%	3.8 ± 0.69	71%
				Potamogeton crispus	Curly pondweed	POCR3	1	0%	tr	0%
				Potamogeton nodosus	Longleaf pondweed	PONO2	1	0%	tr	0%
				Potamogeton spp.	Pondweed	POTAM	5	2%	tr	0%
				Stuckenia pectinata	Sago pondweed	STPE15	7	2%	tr	0%
				-----	Unknown	UNKWN	4	1%	0.2 ± 0.15	3%

Table 6. (Continued)

Lake	Number of Samples	Mean Submerged Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Hendricks	396	14 ± 0.3	4.79							
				-----	Filamentous algae	ALGA	88	22%	0.4 ± 0.08	3%
				Ceratophyllym demersum	Coontail	CEDE4	264	67%	6.1 ± 0.63	45%
				Chara spp.	Muskgrass	CHARA	17	4%	0.2 ± 0.05	1%
				Elodea canadensis	Canadian waterweed	ELCA7	281	71%	5.9 ± 0.58	43%
				Najas guadalupensis	Southern waternymph	NAGU	15	4%	tr	0%
				Potamogeton crispus	Curly Pondweed	POCR3	26	7%	0.3 ± 0.08	2%
				Potamogeton nodosus	Longleaf pondweed	PONO2	3	1%	tr	0%
				Potamogeton spp.	Pondweed	POTAM	82	21%	0.7 ± 0.27	5%
				Stuckenia pectinata	Sago pondweed	STPE15	9	2%	tr	0%
				Zannichellia palustris	Horned pondweed	ZAPA	1	0%	tr	0%
Meadow	244	0.3 ± <0.1	4.59							
				-----	Filamentous algae	ALGA	11	5%	0.1 ± 0.03	28%
				Najas flexilis	Nodding waternymph	NAFL	1	0%	tr	2%
				Najas guadalupensis	Southern waternymph	NAGU	23	9%	0.1 ± 0.03	52%
				Najas minor	Brittle waternymph	NAMI	6	2%	tr	6%
				Stuckenia pectinata	Sago pondweed	STPE15	4	2%	tr	4%
				Zannichellia palustris	Horned pondweed	ZAPA	1	0%	tr	1%
				-----	Unknown	UNKWN	5	2%	tr	8%

Table 6. (Continued)

Lake	Number of Samples	Mean Submerged Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Mormon Trail	408	3 ± 0.1	6.95							
				-----	Filamentous algae	ALGA	61	15%	0.1 ± 0.03	4%
				Ceratophyllym demersum	Coontail	CEDE4	79	19%	1.1 ± 0.33	35%
				Chara spp.	Muskgrass	CHARA	77	19%	0.8 ± 0.20	26%
				Elodea canadensis	Canadian waterweed	ELCA7	75	18%	0.5 ± 0.16	17%
				Isoetes spp.	Quillwort	ISOET	1	0%	tr	0%
				Najas guadalupensis	Southern waternymph	NAGU	113	28%	0.5 ± 0.10	17%
				Potamogeton crispus	Curly pondweed	POCR3	3	1%	tr	0%
				Potamogeton nodosus	Longleaf pondweed	PONO2	7	2%	0.1 ± 0.03	2%
				Potamogeton spp.	Pondweed	POTAM	3	1%	tr	0%
				Stuckenia pectinata	Sago pondweed	STPE15	11	3%	tr	0%
				-----	Unknown	UNKWN	2	0%	tr	0%
Pleasant Creek	612	8 ± 0.2	5.73							
				-----	Filamentous algae	ALGA	64	10%	1.88 ± 0.38	24%
				Ceratophyllym demersum	Coontail	CEDE4	5	1%	0.09 ± 0.07	1%
				Chara spp.	Muskgrass	CHARA	11	2%	0.07 ± 0.04	1%
				Najas guadalupensis	Southern waternymph	NAGU	6	1%	0.02 ± 0.01	<1%
				Najas minor	Brittle waternymph	NAMI	164	27%	3.28 ± 0.47	43%
				Potamogeton foliosus	Leafy pondweed	POFO3	1	0%	tr	<1%
				Potamogeton nodosus	Longleaf pondweed	PONO2	44	7%	1.01 ± 0.22	13%
				Potamogeton pusillus	Small pondweed	POPU7	4	1%	tr	<1%
				Stuckenia pectinata	Sago pondweed	STPE15	68	11%	0.4 ± 0.12	5%
				Vallisneria americana	American eelgrass	VAAM3	34	6%	0.91 ± 0.27	12%
				Zannichellia palustris	Horned pondweed	ZAPA	7	1%	0.01 ± tr	<1%

Table 6. (Continued)

Lake	Number of Samples	Mean Submerged Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Red Haw	505	15 ± 0.6	1.49							
				-----	Filamentous algae	ALGA	12	2%	0.07 ± 0.03	<1%
				Ceratophyllym demersum	Coontail	CEDE4	337	67%	13.7 ± 1.02	89%
				Chara spp.	Muskgrass	CHARA	7	1%	0.36 ± 0.18	2%
				Potamogeton nodosus	Longleaf pondweed	PONO2	74	15%	1.21 ± 0.28	8%
Silver	245	1 ± 0.1	3.85							
				-----	Filamentous Algae	ALGA	31	13%	0.19 ± 0.06	17%
				Ceratophyllym demersum	Coontail	CEDE4	40	16%	0.78 ± 0.28	68%
				Potamogeton pusillus	Small pondweed	POPU7	18	7%	0.07 ± 0.02	6%
				Stuckenia pectinata	Sago pondweed	STPE15	28	11%	0.11 ± 0.02	9%
Smith	310	0.5 ± <0.1	2.04							
				-----	Filamentous algae	ALGA	35	11%	0.4 ± 0.09	90%
				Ceratophyllym demersum	Coontail	CEDE4	1	<1%	tr	1%
				Potamogeton pusillus	Small pondweed	POPU7	6	2%	tr	5%
				Potamogeton spp.	Pondweed	POTAM	1	<1%	tr	4%
				Stuckenia pectinata	Sago pondweed	STPE15	1	<1%	tr	1%

Table 6. (Continued)

Lake	Number of Samples	Mean Submerged Vegetation Abundance Index (%)	Shannon Index	Scientific Species Name	Common Name	Code	N	% Frequency of Occurrence	Abundance Index (%)	% Species Composition of overall abundance
Swan	253	4 ± 0.3	3.71							
				-----	Filamentous algae	ALGA	59	23%	2.7 ± 0.67	63%
				Ceratophyllym demersum	Coontail	CEDE4	2	1%	tr	<1%
				Elodea canadensis	Canadian waterweed	ELCA7	71	28%	1.3 ± 0.39	31%
				Najas flexilis	Nodding waternymph	NAFL	1	<1%	tr	<1%
				Potamogeton foliosus	Leafy pondweed	POFO3	12	5%	0.1 ± 0.04	2%
				Potamogeton pusillus	Small pondweed	POPU7	35	14%	0.2 ± 0.05	4%
Three Fires	154	0.6 ± 0.1	1.97							
				-----	Filamentous algae	ALGA	13	8%	0.5 ± 0.17	72%
				Ceratophyllym demersum	Coontail	CEDE4	3	2%	0.1 ± 0.07	17%
				Potamogeton spp.	Pondweed	POTAM	1	1%	0.1 ± 0.07	10%
Wapello	702	8 ± 0.2	3.43							
				-----	Filamentous algae	ALGA	10	1%	0.1 ± 0.04	1%
				Ceratophyllym demersum	Coontail	CEDE4	362	52%	6.6 ± 0.55	87%
				Chara spp.	Muskgrass	CHARA	8	1%	tr	<1%
				Elodea canadensis	Canadian waterweed	ELCA7	41	6%	0.3 ± 0.11	4%
				Heteranthera dubia	Water star-grass	HEDU2	32	5%	0.2 ± 0.04	2%
				Najas flexilis	Nodding waternymph	NAFL	4	1%	tr	<1%
				Najas minor	Brittle waternymph	NAMI	10	1%	tr	<1%
				Potamogeton crispus	Curly pondweed	POCR3	5	1%	tr	<1%
				Potamogeton nodosus	Longleaf pondweed	PONO2	42	6%	0.2 ± 0.05	3%
				Potamogeton pusillus	Small pondweed	POPU7	6	1%	0.1 ± 0.04	1%
				Stuckenia pectinata	Sago pondweed	STPE15	7	1%	tr	<1%

Table 7. Summary of nMDS r^2 and P-value of the physical-chemical parameters in 13 Iowa Lakes in 2007. P-value is based on 1000 permutations. Statistically significant relationships of vegetation abundance (i.e., emergent/floating and submerged) and physical-chemical parameters are determined by ($P \le 0.05$). Significant values are in bold.

Physical-chemical parameters	Emergent/Floating Aquatic Vegetation		Submerged Aquatic Vegetation	
	r^2	P-value	r^2	P-value
Alkalinity	0.49	0.10	0.38	0.21
Chlorophyll *a*	0.50	0.09	0.53	0.08
Hardness	0.40	0.18	0.30	0.34
pH	0.58	**0.04**	0.06	0.91
Secchi-depth	0.43	0.13	0.62	**0.03**
Temperature	0.43	0.14	0.11	0.08
TKN :TP	0.28	0.36	0.40	0.19
Total Kjeldahl Nitrogen (TKN)	0.57	**0.04**	0.43	0.15
Total phosphorus (TP)	0.30	0.33	0.44	0.15
Total suspended solids (TSS)	0.39	0.18	0.50	0.08

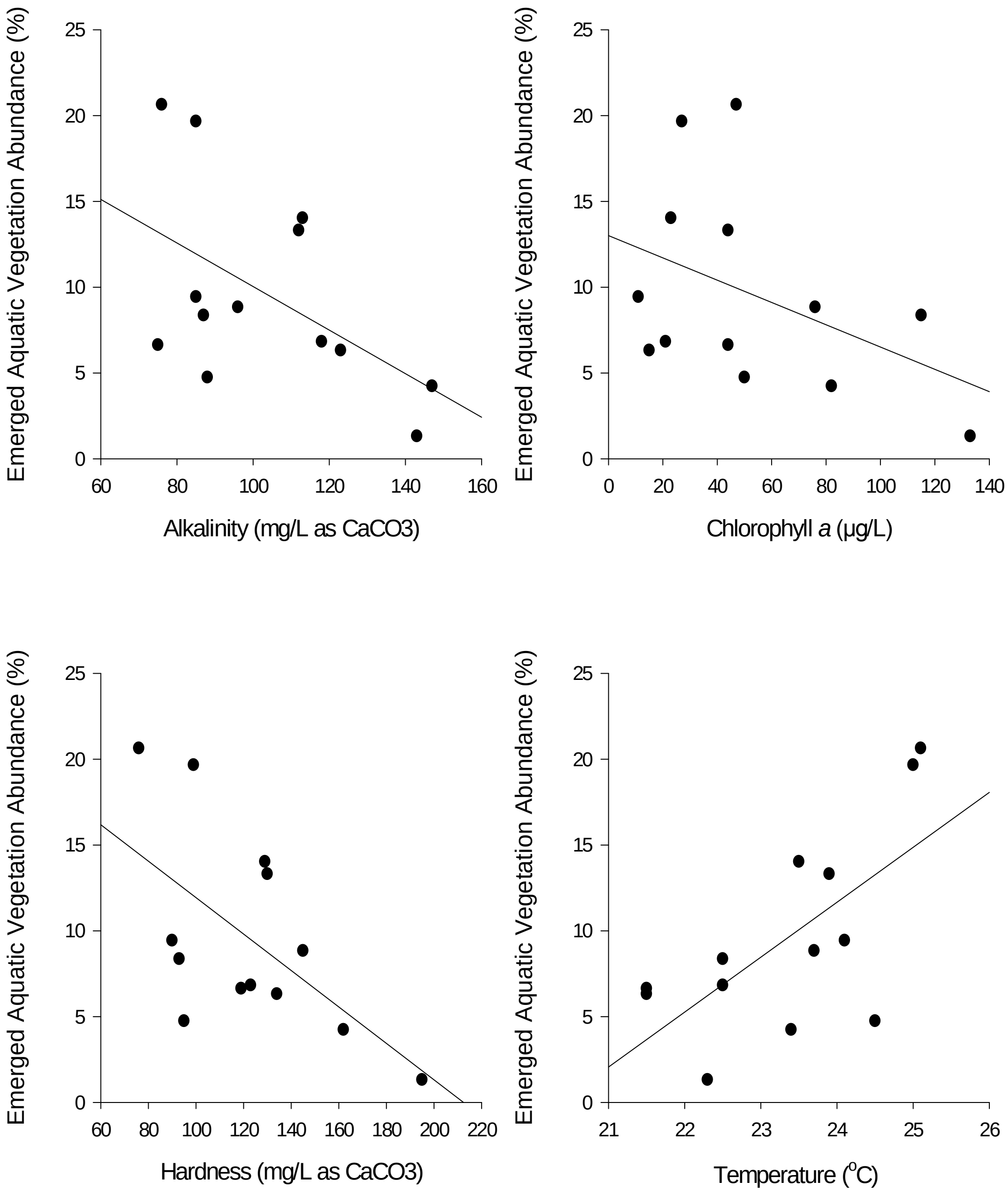

Figure 1. Summary of simple linear regression plots between arcsine (square root) transformed emergent/floating vegetation abundance and mean physical-chemical parameters (i.e, alkalinity, chlorophyll *a*, hardness, and temperature) in 13 Iowa lakes in 2007. Statistically significant relationships between emergent/floating vegetation abundance and mean environmental parameters are determined by ($P \leq 0.05$).

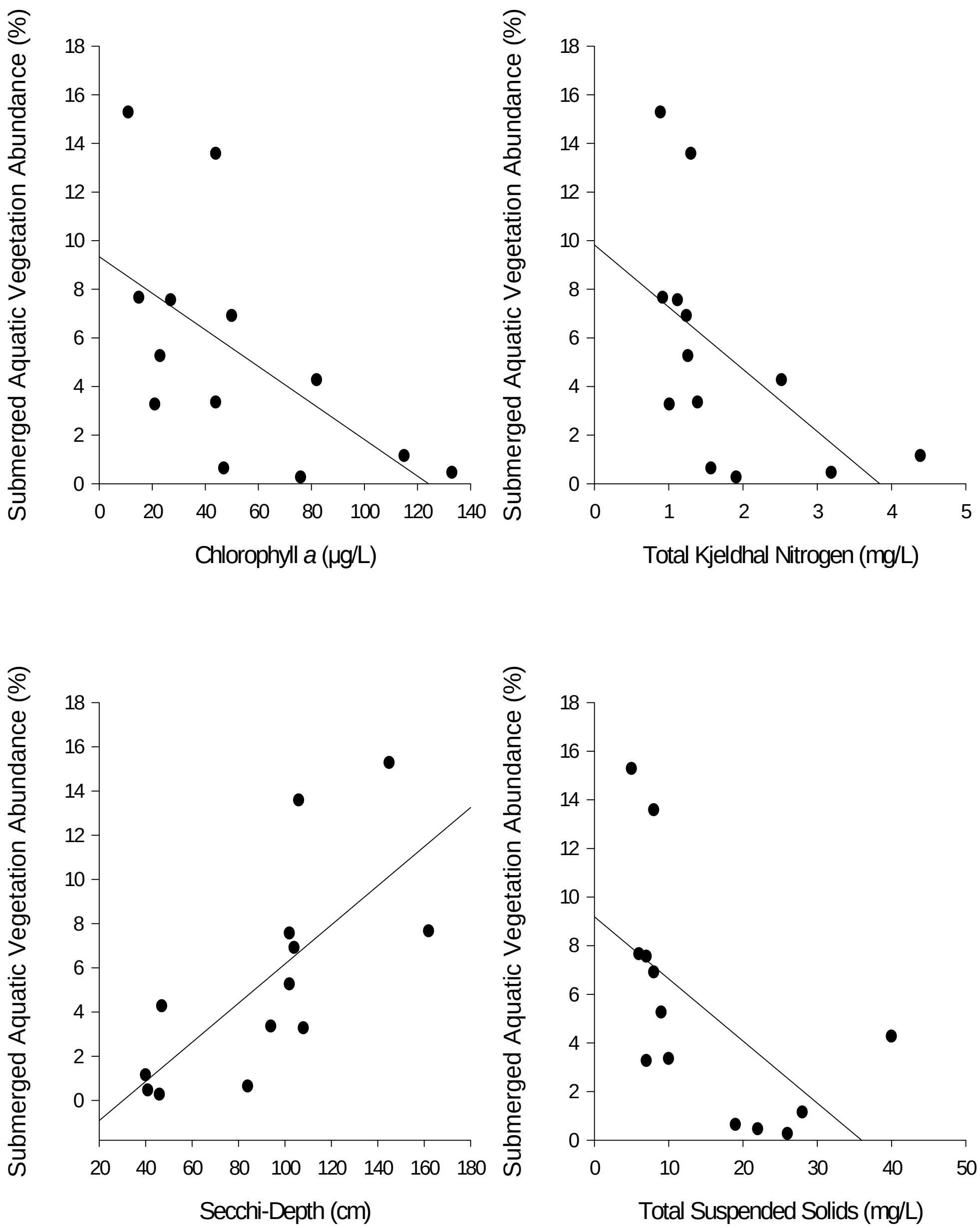

Figure 2. Summary of simple linear regression plots between arcsine (square root) transformed submerged vegetation abundance and mean physical-chemical parameters (i.e., chlorophyll a, Secchi-depth, total Kjeldahl nitrogen, and total suspended solids) in 13 Iowa lakes in 2007. Statistically significant relationships between submerged vegetation abundance and mean environmental parameters are determined by ($P \leq 0.05$).

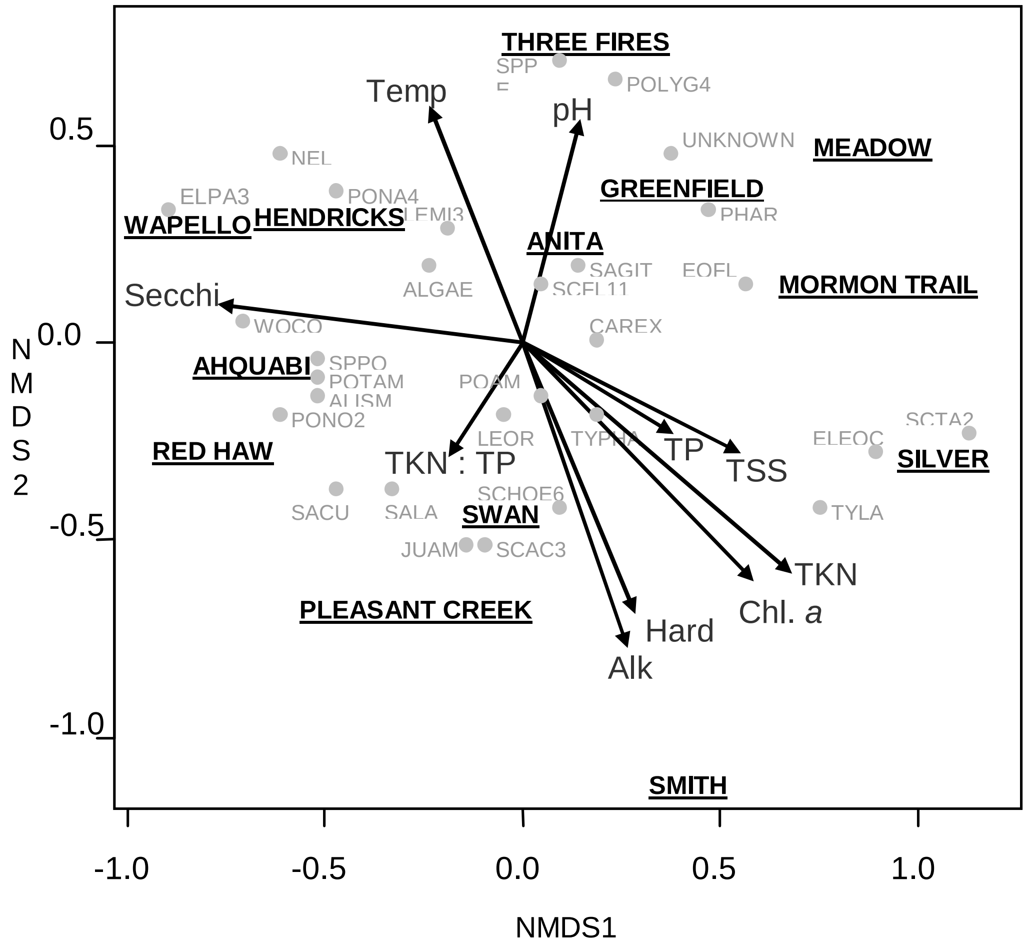

Figure 3. Non-metric multidimensional scaling (NMDS) plot illustrating the strength and relationship among physical-chemical parameters (vectors), lakes (underlined), and emergent/floating aquatic vegetation (gray) in 13 Iowa lakes in 2007. Plant vegetation codes are located in Table 5.

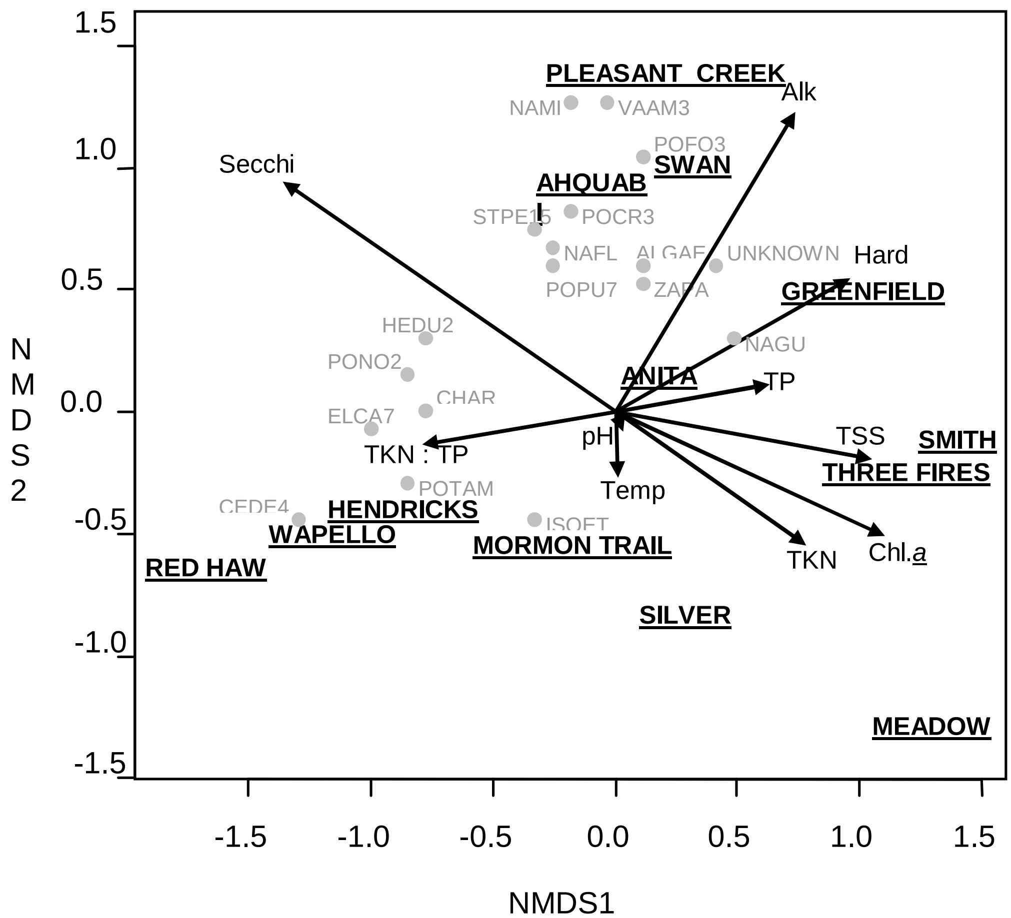

Figure 4. Non-metric multidimensional scaling (NMDS) plot illustrating the strength and relationship among environmental parameters (vectors), lakes (underlined), and submerged aquatic vegetation (gray) in 13 Iowa lakes in 2007. Plant vegetation codes are located in Table 6.

CHAPTER 3. LITTORAL INFLUENCES ON ZOOPLANKTON POPULATIONS AND JUVENILE BLUEGILLS IN IOWA LAKES

A paper to be submitted to *North American Journal of Fisheries Management*

MEGAN A. ERNST AND JOSEPH E. MORRIS

Department of Natural Resource Ecology and Management
Iowa State University
Ames, Iowa, USA 50011

ABSTRACT

Aquatic vegetation helps maintain the overall integrity of aquatic ecosystems. Lakes with vegetation are characteristic of reduced chlorophyll concentrations, lower phytoplankton densities, and large-bodied cladocerans. Littoral zones with dense vegetation beds accommodate invertebrate communities that are richer in abundance and diversity compared to barren littoral zones. Two objectives of this research were to determine whether vegetated and non-vegetated littoral zones have similar zooplankton populations, and the role of the littoral zone upon juvenile bluegills food habits. Vegetation-loving cladocerans, e.g., *Chydorus* spp., were typically found in higher abundance in vegetated areas compared to open littoral and limentic zones, while limnetic zooplankton, *Daphnia* spp. was often found in higher concentrations in pelagic zone. Regardless of fish size (≤50mm and >50mm), prey selectivity was similar. However, different sampling periods (spring/summer vs. fall) showed different prey choices.

INTRODUCTION

Aquatic vegetation plays a vital role in maintaining the overall integrity of aquatic ecosystems. Vegetation stabilizes aquatic ecosystems by reducing nutrient concentrations (van Donk et al. 1989) and shoreline erosion, providing food and habitat for aquatic fauna, and increasing water clarity, producing oxygen, reducing shore erosion. (Canfield et al. 1984; Timms and Moss 1984; Jeppesen et al. 1990; Scheffer et al. 1993; Meijer et al. 1994; Moss et al. 1994; Egertson et al. 2004). Aquatic vegetation abundance is influenced by factors such as irradiance, temperature, water chemistry (nitrogenous and phosphorus nutrients), wave action, lake size, and catchment basin morphology (Gasith and Hoyer 1998).

In addition to the previously described physical effects of the environment on aquatic plants, their presence can have a positive effect on zooplankton biomass and a negative effect on phytoplankton biomass (van Donk and van de Bund 2002). Submerged vegetation provides refuge for algae-eating zooplankton (e.g., cladocerans escapement from zooplankitvorous fish; Timms and Moss 1984). Aquatic vegetation also indirectly contributes to fish growth and recruitment by increasing and diversifying invertebrate communities as well as providing age-0 fish with protection from predation by reducing some predators’ visibility and maneuverability (Savino and Stein 1982). Dense macrophytes can actually serve as refugees for large-bodied cladocerans escaping from predation from zooplanktivorous fish which is consistent with the refugee hypothesis for grazing zooplankton (Timms and Moss 1984; Stansfield et al. 1997).

Aquatic vegetation abundance increases invertebrate biomass by 5 to 24% (Wiley et al. 1984). Timms and Moss (1984) determined lakes with vegetation have characteristics of reduced chlorophyll *a* concentrations, lower phytoplankton densities, and more large-bodied cladocerans. Littoral zones with dense vegetation beds accommodate invertebrate communities that are richer in abundance and diversity compared to barren littoral zones. (Eadie and Keast 1984; Diehl 1988; Engel 1988; Bryan and Scarnecchia 1992). Aquatic vegetation provides a refuge for algae-eating zooplankton against fish predation (Timms and Moss 1984), while periphyton, the layer covering macrophytes and decomposing plants, are important food sources for invertebrates (Engel 1998).

Members of the Centrarchidae family are well known inhabitants of littoral zones of lakes. Centrarchids have been shown to use vegetated zones for protection against predation and use non-vegetated zones to maximize foraging efficiency on visible zooplankton (Werner et al. 1983; Werner and Hall 1988). Given the importance of vegetated habitats to centrarchids, major population changes can result from an abrupt reduction of vegetation abundance. As an example, following a drastic macrophyte removal in Lake Conroe, Texas, algal biomass increased rapidly, cyanobacteria dominated summer blooms, water clarity decreased, and bluegill *Lepomis macrochirus* and major zooplankton taxa biomass decreased (Bettoli et al. 1993).

In the majority of lakes, reservoirs, and ponds, zooplankton group (e.g., copepods, cladocerans, and rotifers) populations are in a constant state of flux (Pennak 1966). Zooplankton populations can respond quickly to environmental changes, making them a good trophic condition indicator (Gannon and Stemberger 1978). Zooplankton are found throughout all lake types but certain species, especially rotifers, may indicate extreme trophic states (Gannon and Stemberger 1978). Oligotrophic lakes have small invertebrate biomass composed of a variety of zooplankton species while eutrophic lakes exhibit a large biomass with fewer species (Gannon and Stemberger 1978). A combination of low flushing rates, abundance of nutrients and food source, and low levels of toxicants and algal metabolites may provide the best conditions for a large and diverse zooplankton community.

Aquatic vegetation provides habitat structure for dense phytoplankton-consuming zooplankton populations and inhibits phytoplankton production by shading waters and sequestering nutrients. As a result algal density is low and only small fast growing algae and bacterioplankton survive (Schriver et al. 1995). Submerged aquatic vegetation beds with low phytoplankton populations are typically unfavorable foraging habitat for pelagic zooplankton, but during the day due to the reduced predation risk, zooplankton are typically found aggregating near the edge of these beds (Davies 1985; Paterson 1993; Scheffer 2004). Scheffer (2004) described zooplankton, .e.g., *Daphnia* spp., migrating to avoid predation. One study showed daytime density of *Daphnia hyaline* and *Daphnia galeata* in a vegetated littoral zone increasing to 1776 L^{-1} from 638 L^{-1} at night in the same littoral zone. Some zooplankton taxa (e.g., *Daphina* spp), respond to environmental chemical cues by changing behaviors (Dodson 1988; Demeester et al. 1995; Stirling 1995), growth rates, and body morphology to avoid predation (Weider and Pijanowska 1993; Engelmayer 1995).

Littoral zones of lakes provide food, cover, and spawning sites for centrarchid fishes (Lemly and Dimmick 1982). Juvenile centrarchids feed primarily in littoral zones (Werner and Hall 1988; Schneider 1999) and use aquatic vegetation as a refuge from piscivorous fish (Crowder and Cooper 1982; Savino and Stein 1982; Mittelbach 1988). Durocher et al. (1984) showed a strong positive relationship between percent submerged aquatic vegetation and largemouth bass *Micropterus salmoides* standing crop and numbers being recruited to harvestable size. Age-0 fish feed primarily on zooplankton species consuming significant amounts of rotifers, small cladocerans (*Bosmina* spp.), and developmental stages of copepods (nauplii) (Mehner and Thiel 1999). Lemly and Dimmick (1982) research compared several lakes over time and indicated that feeding relationships of centrarchids were similar regardless of catch per unit effort and centrarchid species assortment. Age-0 bluegills *Lepomis machrochirus* migrate from littoral to limnetic back to littoral zones within 30-40 days after hatching. This migration is thought to occur to escape predation from aquatic insects (e.g., Corixidae) that inhabit littoral zones. Once young bluegills are large enough, they return to limnetic zones to feed upon the abundant zooplankton (Beard 1982).

Excessive abundance of aquatic vegetation can also hinder a fishery such as largemouth bass (Wiley et al. 1984; Bettoli et al. 1992). Shallow, densely vegetated lakes are most likely to have a high abundance of small, slow-growing bluegills because aquatic vegetation reduces feeding rates by piscivores thus lowering predator-induced mortality rates on small fish (Crowder and Cooper 1982; Savino and Stein 1982; Gotceitas and Colgan 1989). Crowder and Cooper (1982) determined that intermediate aquatic vegetation density enabled bluegills to grow best compared to low and high aquatic vegetation densities.

Density assessments of littoral zooplankton are inherently difficult to determine due to variable habitats (vegetated vs. non-vegetated) in the shallow zone. Plankton nets moved across the surface often produce misleading results by sampling only a portion of the water column (Pennak 1966) or scare away zooplankton from the vegetation area (Downing and Cyr 1985). Also, littoral zooplankton population estimates in vegetation beds are highly variable, requiring numerous replicated samples to reduce large variances and to increase precision (Downing 1986).

The goal of this project is to better understand the role of aquatic vegetation in Iowa lakes. The study objectives are: 1) assess zooplankton population dynamics in vegetated and non-vegetated littoral zones and 2) determined littoral influences on prey selectivity in juvenile bluegills.

STUDY AREA

Aquatic vegetation and environment data were collected monthly May to September 2007, from three lakes (impoundments) located in southeastern Iowa (Figure 1). The three study lakes (Lake Ahquabi, Red Haw Lake, and Lake Wapello) varied in size, depth, and grass carp abundance as well as presence of invasive plants species (e.g., curly leaf *Potamogeton crispus*; Table 1).

METHODS

Water Quality Collections- Water quality samples were collected bi-monthly May to September 2007 in limnetic waters near the dam structure from three lakes, Red Haw Lake (30.63 ha) Lake Ahquabi (47.29 ha) and Lake Wapello (114.33 ha). Sampling points were stationary, chosen randomly and located using a Garmin GPSmap 76CSX Global Positioning System (GPS). Collection was accomplished using an integrated 2-m tube sampler with a one-way check valve. Water quality analysis was completed by University of Iowa Hygienics Laboratory to assess levels of TP, total Kjeldahl nitrogen (TKN), total suspended solids (TSS), chlorophyll *a*, hardness, and alkalinity.

Aquatic Vegetation Collections- Aquatic vegetation surveys were conducted the first week of each month from May to September 2007. Stationary transect lines were randomly selected around the perimeter of each lake. Transects for Red Haw Lake, Lake Ahquabi and Lake Wapello were 13, 19 and 25, respectively using (Quist et al. 2007) as a guideline.

Aquatic vegetation was categorized as either submerged (SAV) or emergent/floating (EFAV) along each transect line. The sampling device for SAV consisted of two welded garden rake heads measuring 35.6 cm in length and having 14, 5.1-cm teeth attached to an extendable 5.5-m push pole (Yin et al. 2000). It was lowered to the substrate, turned 180 degrees, raised, and pulled horizontally through the surface water to rinse and compact aquatic vegetation on the rake head. Total percent coverage was estimated using marked gradations on the teeth; percent species coverage was visually estimated for each sample. Emergent/floating aquatic vegetation was sampled by placing a floating, 1-m diameter hoop on the surface water and overall and species percent coverage were quantified for each sample. The percent coverage of the total sample of a particular species was calculated by multiplying overall percentage by that species coverage and dividing the product by 100. For instance, in a SAV sample that has 50% coverage and two species of plants that comprise of 20 and 80%, the subsequent individual coverage of the total sample is 10 and 40%, respectively.

Transects were sampled perpendicular from the waters edge outward at 0.61-m contour depth increments to a minimum of 2.4-m for both SAV and EFAV samples. Transects were complete when two consecutive rake samples were void of SAV past the 2.4-m mark, when depth contours indicated a decrease in water depth, or when depths reached 4.9-m. If aquatic vegetation was quantifiable at 4.9-m, one last pull occurred at 5.5-m and aquatic vegetation was noted as being present or absent.

Zooplankton Collections- Red Haw Lake had three sampling sites and Lakes Ahquabi and Wapello had four sites. Zooplankton samples were collected bi-monthly from littoral vegetated and non-vegetated areas $\leq$1.2 m depths using a 12-volt DC pump (Simmer Blue Water Pump, Model No. BW85P, Delavan, Wisconsin) with 1.6 cm diameter hose to collect integrated water samples from the entire water column. One 20-L sample filtered through an 80–μm mesh Wisconsin-style net was collect and preserved in 4% buffered formalin. In addition, a box sampler (Downing 1986) was used to better sample zooplankton (i.e., Chydoridae) that are often associated more with aquatic vegetation (Pennak 1966). When a vegetated area was present, vegetation with accompanying water was enclosed in the sampler box, brought to the boat shaken to dislodge zooplankton, rinsed with original box sampler water, filtered through the same 80–µm mesh Wisconsin-style net, and then preserved with 4% buffered formalin. Samples were then returned to the lab for identification and enumeration.

Fish Collections- Juvenile bluegills were sampled in the same location as zooplankton and water quality in June and September, designated spring/summer and fall collections, respectively. A hand-held DC probe was used while standing in a boat in water depths $\leq$1.2 m. The actual voltage and amperage varied depending on local conditions. Fifteen-minute electrofishing runs were completed parallel to the shore, placing the GPS point in the middle of the run. After each run total length and weight of the first 50 fish/species were recorded. If available, ten bluegills were killed per site with Finquel® and then placed in 10% buffered formalin for later stomach analysis.

Fish Stomach Analysis- Total length (TL) and weight of preserved bluegills were recorded prior to dissection. Stomach contents were identified and enumerated. Zooplankton were identified as cladocera (genera), rotifera, ostracoda, cyclopoida, calanoida, copepod, and nauplii. Aquatic insects and prey fish were identified to the lowest possible taxa (Michaletz et al. 1987).

Statistical Analysis- Mean zooplankton densities were log transformed to achieve normality. To compare vegetated and non-vegetated littoral zones, a one-way analysis of variance (ANOVA) was computed using JMP® 7.0.2, a statistical software package of SAS Institute (2007). A Bonferroni correction level of .0033 was used to determine statistical significance. A Tukey-test was used to determine which sampling method/location was significantly different from each other.

Average abundances of zooplankton genera were compared to juvenile bluegill stomach contents to determine relationships among zooplankton populations and bluegill preferences in foraging locations and prey selectivity. Frequency of occurrence and composition by number were calculated. Frequency of occurrence (%-O) is the proportion of the bluegills' stomachs containing a specific family or genus (e.g., *Chydorus* spp.) while composition by number (%-N) is the number of prey items in a specific family or genus relative to the total number of prey consumed.

A linear index of food selection (Strauss 1979) was calculated to determine prey selection by the captured bluegill. Relative abundance of specific prey items were estimated using mean %-N values for each station using pump samples from both vegetative and non-vegetative samples. The sample dates used in this calculation were samples collected from 30 d prior to up to actual sample date of the fish collection. The non-parametric sign test was used to determine if the indices were significant different from 0.

RESULTS

The three lakes had similar environmental conditions (water temperature, pH, TP, TKN, TKN:TP, TSS, alkalinity and hardness) throughout the 2007 sampling season with the exception of Secchi-depth and chlorophyll *a* levels in Lake Red Haw (Table 2). Lake Red Haw's average Secchi-depth of 145±19 cm and chlorophyll *a* average of 11±2 µg/L (Table 2) are two measurements showing good light penetration, possibly reflective of the increased abundance of submerged vegetation (See Ernst et al. 2008).

Aquatic vegetation influences zooplankton genera species and abundance differently (Table 3). Cladocerans, in particular the Chydoridae family, were typically found in higher abundance in vegetated areas compared to open littoral and limentic zones, while limnetic zooplankton, *Daphnia* spp. was typically found in higher concentrations in pelagic zone (Table 3). Lake Red Haw had the highest density of submerged aquatic vegetation and the largest *Chydorus* populations (box = 898 ± 892 L^{-1}, veg = 120 ± 6 L^{-1}) sampled from littoral vegetation (Table 3). Ostracods abundance varied among the sampling methods and was found most abundant in littoral vegetated zones (Table 3). A one-way ANOVA (Bonferroni correction = 0.0033) comparing non-vegetated littoral zone (open), vegetated littoral zone (veg), a vegetated littoral zone using a box sampler (box), and limnetic zone for a reference indicated significant relationships for certain taxa of zooplankton (Table 4). When significant relationships were determined for individual genera of zooplankton, generally box and vegetation samples were statistically different from open and limnetic samples, but similar to each other.

Juvenile bluegills are often found in vegetated littoral areas and their stomach content indicated that they also fed there. *Chydorus* spp. were in high abundance in vegetated areas and were also often found in centrarchids' stomachs, indicating a possible relationship between prey and predator habitats (Tables 5 and 6). At ca. 50 mm TL, bluegills switch from a diet of zooplankton to aquatic insects (1977). This is illustrated by chironomids, the prey of choice amongst both size groups across lakes and sampling periods (Tables 5 and 6). In the spring/summer, the three major prey items fed upon by juvenile bluegills most were *Chydorus* spp., ostracods, and chironomids (Tables 5 and 6). There is little difference between food habits of juvenile centrarchids regardless of size group (Tables 5 and 6).

In Lake Ahquabi, *Chydorus* spp. were heavily preyed upon as well as ostracods, while in Lake Red Haw, *Chydorus* spp. and aquatic insects were the food choice in the spring sampling period; amphipods became an important food in the fall (Tables 5 and 6). Lake Wapello bluegills primarily feed on cladocerans and chironomids in both spring and fall. Vegetation did influence zooplankton species and abundance (Tables 3 and 4), as well as influencing prey selectivity (e.g., *Chydorus*) in juvenile bluegills.

Strauss prey selectivity index shows that lakes with higher submerged aquatic vegetation (Lake Red Haw and Lake Wapello) contain bluegills that are less selective and more opportunistic in feeding habits (Tables 7 and 8). Vegetation provides habitat for zooplankton, enabling high densities of chydorids to inhabit these waters.

DISCUSSION

Submerged aquatic vegetation protects zooplankton against fish predation, as well as prey fish from piscivorous fish; protection effectiveness depends on strand densities versus predator density (Savino and Stein 1982; Scheffer 2004).

Submerged vegetation also increases complexity of littoral zones, providing habitat to a variety of zooplankton species (Campbell et al. 1985). Cladocerans, especially from the family of Chydoridae, have been shown to have high population abundance in plant beds (Scheffer 2004). Chydorids have appendages and behaviors that are adapted to living on plant surfaces, and are difficult to dislodge even when disturbed (Pennak 1966; Fryer 1968; Campbell et al. 1985).

Sommer and Stibor (2002) determined that calanoid copepods populations peaked in oligotrophic lakes, *Daphnia* in mesotrophic lakes and cyclopoid copepods in eutrophic lakes. While all three lakes have similar TP and TKN averages and overall low calanoid copepod populations, Lake Wapello had the highest density (7 ± 3 L^{-1}) found in the limnetic zone sample. Red Haw Lake with the lowest TP and TKN values (0.016±0.002 mg/L and 0.889 ± 0.070 mg/L) produced the largest *Daphnia* population (13 ± 9 L^{-1}) in the limentic zone. The vegetated littoral zone within Red Haw Lake produced the highest density of cyclopoid copepod (236 ± 79 L^{-1}).

The three lakes are considered as being eutrophic and share similar physical-chemical parameters with the exception of Secchi-depth and chlorophyll *a* levels in Lake Red Haw. Ernst et al. (2008) noted that these two variables had a significant relationship with overall aquatic vegetation abundance. Lake Red Haw had the largest chydorid populations as well as the highest submerged aquatic vegetation abundance (15.28 ± 0.595%) consisting primarily of a dense population of coontail (*Ceratophyllum demersum*; 89% of the vegetation present). Chilton (1990) research determined that coontail with dissected leaves often supported higher density of invertebrates, compared to simple leaved plants like american eelgrass (*Vallisneria* spp.). My research agrees with many other studies in suggesting that the family Chydoridae (i.e., *Alona* spp., *Alonella* spp., and *Chydorus* spp.) has an affinity towards submerged aquatic vegetation. The genus *Chydorus* spp. (898 ± 892 L^{-1}), *Alona* spp. (258 ± 196 L^{-1}), *and Alonella* spp. (57 ± 43 L^{-1}) had much larger populations within vegetation (box sample) than in open littoral zones (4 ± 1 L^{-1}, 1 ± tr L^{-1}, 7 ± 6 L^{-1}). Lake Wapello, has the most dense emergent/floating vegetation abundance (19.66± 0.431%), but a much lower chydorid population than Red Haw Lake, possible suggesting chydorids need submerged vegetation as habitat rather than emergent/floating vegetation (Table 3).

Sampling method choice is important depending on target zooplankton species. Sample collectors should focus on sampling littoral vegetated zones rather than non-vegetated littoral or limnetic areas to adequately assess zooplankton densities living within vegetation (Chydoridae family). Most densities of zooplankton taxa were significantly different depending on sampling method. The statistical analyses frequently revealed that the box sampler and pump produced similar population densities while open littoral and limnetic zones were similar. Results show cladocerans that are often associated with vegetation (*Chydorus* spp. and *Alona* spp.), are more abundant in the littoral vegetation and box sample. Copepods, especially nauplii (655 ± 300 L^{-1}), were also in high abundance in vegetated areas of Red Haw Lake. The nauplii as well as other zooplankton genus may take refuge from zooplanktivorous fish (Timms and Moss 1984) in the heavily abundant coontail due to Red Haw Lake's low chlorophyll *a* levels (11 ± 2 μg/L) and high Secchi-disk transparences (145 ± 19 cm). Wiley et al. (1984) determined that zooplankton abundance increased with aquatic vegetation densities; however, this may represent a concentration of zooplankton in vegetation beds, and not a real increase in overall zooplankton populations throughout the lake.

Cladocerans are the dominant zooplankton in many lakes (Sommer and Stibor 2002). In the three study lakes, *Chydorus* spp. and *Alona* spp. were the dominant cladoceran genus, while nauplii was the most abundant copepod. Cladocerans have simple life cycles consisting of parthenogentic reproduction throughout most of the year without larval stages and becoming sexual mature within a few days at water temperatures of 20^{o}C (Sommer and Stibor 2002). Unlike copepods, cladocerans still grow after maturity, making them a prime target to zooplanktivories fish (Sommer and Stibor 2002), possibly explaining the reduced levels of many cladoceran genera (i.e., *Daphnia* spp. and *Ceriodaphnia* spp.) in my lakes. In contrast, copepods have a more complicated life cycle (nauplius stages, and subadult stages) and slow growth leading to long generation cycles (1 month until maturity) and low birth rates (Sommer and Stibor 2002). The long generation cycle and low birth rates may explain why nauplii stage was the most abundant copepod in the three study lakes.

Stomach contents may not accurately portray centrarchids' diets due to rapid digestion of some prey selections (Lagler 1956; Lemly and Dimmick 1982). Another disadvantage of using stomach contents, is that slow digesting contents may accumulate more heavily (e.g., chironomids) and may not reflect true proportions of ingested prey (Lemly and Dimmick 1982). My research focused only on zooplankton; therefore, I can only note presence and numbers of aquatic insects found in the stomach without relating to environmental conditions. However, chironomid populations are often prevalent in littoral zones of lakes.

Lemly and Dimmick (1982) research showed post-larval bluegills eat more copepods than other zooplankton groups and that this gradually switches to insect larvae. Schneider (1999) found that bluegills in Michigan waters primarily foraged on *Daphnia* spp., *Chaoborus* spp., and chironomids. Also, bluegills ranging in size 25-75 mm ate small midge larvae, mayfly nymphs, and copepods, while bluegills 75-125 mm feed on larger size caddisflies (Trichoptera) and damselflies (Zygoptera). My research results showed that regardless of size, bluegills ate *Chydorus* spp., chironomids, and ostracods, but did not necessarily active select for these taxa.

Dewey et al. (1997) determined young bluegill fed on small prey, e.g., chydorids, *Daphnia*, and other cladocera, while adult bluegills consumed amphipods, gastropods, and odonates; both life stages consumed chironomids (Dewey et al. 1997). Regardless of fish size (≤50mm and >50mm), prey selectivity was similar; however, different sampling periods (spring/summer vs. fall) showed different prey choices. In Lake Ahquabi, ca. 90% of ≤50mm bluegill's diet during the spring/summer and fall sampling period consisted of 64% *Chydorus* spp. (Table 5). Small bluegills (<50 mm) in Red Haw Lake, the lake with the highest submerged aquatic vegetation and best Secchi-reading of the study lakes, fed on chironomids (35% of diet) and amphipods in both spring and summer (14% of diet). During the fall sampling period, bluegills feeding habits shifted from amphipods (2% of diet) to *Chydorus* spp. (23% of diet).

Seasonal variation in bluegill diets may have, in part, to do with life history stages of prey (e.g., aquatic insects; Mittelbach 1981). In the fall, larval aquatic insect populations are low, because most have matured and emerged as aerial adults (Mittelbach 1981). During this seasonal change, juvenile fish feed primarily on chydorids, gastropods, amphipods, *Daphnia* spp., and, *Bosmina* spp. (Dewey et al. 1997). In all three of the study lakes, chironomids as a food source decreased from spring/summer food habits to fall food habits supporting Mittelbach (1981) and Dewey et al. (1997) findings.

In Lake Red Haw, it is worth noting that abundance of *Chydorus* spp. population in the vegetated zooplankton samples resulted in limited significant values of prey selection indices compared to the other two lakes. One possible reason is the abundance of higher energetic food source, e.g., chironomids. Another possibility is that the dense coontail provided refuge to *Chydorus* spp. and hindered the bluegills ability to locate them until vegetation died back in the fall bluegill sampling period. Bluegills, being opportunistic feeders, most likely did little prey selection, rather ate what was most abundant and in the case of Red Haw Lake, it was the Chydoridae family.

Management Implications

When studying multiple lakes, the sampling method of using a water pump in a vegetated zone may be the most effective method in reference to time management. Using the box sample and vegetated sample, most zooplankton genera population differences were insignificant. If sampling for a particular genus (i.e., *Chydorus*) a box sampler may prove to be a better method, producing higher population numbers.

Juvenile bluegills consume more aquatic insects than initially thought. By sampling for only zooplankton, a main component of bluegill food habitats was overlooked. Future research should incorporate littoral benthic samples to determine the ratio of prey present, e.g., zooplankton, aquatic insects, to choice of prey consumed. In addition to fully understand how juvenile bluegills and other centrarchids are using aquatic vegetation for feeding habitats, lakes void of vegetation need to be incorporated into this study to determine zooplankton taxa and densities, and related centrarchid prey selection in those lakes.

REFERENCES

Beard,T.D. 1982. Population dynamics of young-of-the-year bluegill. Wisconsin Department of Natural Resources Technical Bulletin Number 127, Madison, Wisconsin, USA.

Bettoli, P.W., M.J. Maceina, R.L. Noble, and R.K. Betsill. 1992. Piscivory in largemouth bass as a function of aquatic vegetation abundance. North American Journal of Fisheries Management 12:509-516.

Bettoli, P.W., M.J. Maceina, R.L. Noble, and R.K. Betsill. 1993. Response of a reservoir fish community to aquatic vegetation removal. North American Journal of Fisheries Management 13:110-124.

Bryan, M.D. and D.L. Scarnecchia. 1992. Species richness, composition, and abundance of fish larvae and juveniles inhabiting natural and developed shorelines of a glacial Iowa lake. Environmental Biology of Fishes 35:329-341.

Campbell, J.M, J.E. Morris, and R. L. Noble. 1985. Spatial variability and community structure of littoral microcrustacea in Lake Conroe, Texas. The Texas Journal of Science 36:247-256.

Canfield, D.E. Jr., J.V. Shireman, D.E. Colle, W.T. Haller, C.E. Watkins II, and M.J. Maceina. 1984. Prediction of chlorophyll *a* concentrations in Florida lakes: importance of aquatic macrophytes. Canadian Journal of Fisheries and Aquatic Sciences 41:487-501.

Carlander, K.D. 1977. Handbook of freshwater fishery biology. Volume 2. Iowa State University Press, Ames, IA, U.S.A.

Chilton, E.W. II. 1990. Macroinvertebrate communities associated with three aquatic macrophytes (*Ceratophyllum demersum, Myriophyllum spicatum, and Vallisneria Americana)* in Lake Onalaska, Wisconsin. Journal of Freshwater Ecology 5:455-466.

Crowder, L.B. and W.E. Cooper. 1982. Habitat structural complexity and the interaction between bluegills and their prey. Ecology 63:1802-1813.

Davies, J. 1985. Evidence for a diurnal horizontal migration in *Daphnia hyalina lacustris* Sars. Hydrobiologia 120:103-105.

Demeester, L., L.J. Weider, and R. Tollrian. 1995. Alternative antipredator defenses and genetic polymorphism in pelagic predator-prey system. Nature 378: 483-485.

Dewey, M.R., W.B. Richardson, and S.J. Zigler. 1997. Patterns of foraging and distribution of bluegill sunfish in a Mississippi River backwater: influence of macrophytes and predation. Ecology of Freshwater Fish 6:8-15.

Diehl, S. 1988. Foraging efficiency of three freshwater fishes effects of structural complexity and light. Oikos 53:207-214.

Dodson, S. 1988. The ecological role of chemical stimuli for the zooplankton predator-avoidance behavior in *Daphina*. Limnology and Oceanography 33:1431-1439.

Downing, J.A. 1986. A regression technique of the estimation of epiphytic invertebrate populations. Freshwater Biology 16:161-173.

Downing, J.A. and H. Cyr. 1985. Quantitative estimation of epiphytic invertebrate populations. Canadian Journal of Fisheries and Aquatic Science 42:1570-1579.

Durocher, P.P., W.C. Provine, and J.E. Kraai. 1984. Relationship between abundance of largemouth bass and submerged vegetation in Texas reservoirs. North American Journal of Fisheries Management 4:84-88.

Eadie, J. M., and A. Keast. 1984. Resource heterogeneity and fish species diversity in lakes. Canadian Journal of Zoology 62:1689-1695.

Egertson ,C. J., J.A. Kopaska, and J.A. Downing. 2004. A century of change in macrophyte abundance and composition in response to agricultural eutrophication. Hydrobiologia 524:145-156.

Engel, S. 1988. The role and interactions of submersed macrophytes in a shallow Wisconsin Lake USA. Journal of Freshwater Ecology 4:329-342.

Engelmayer, A. 1995. Effects of predator-released chemicals on some life history parameters of *Daphnia pulex*. Hydrobiologia 307:203-206.

Fryer, G. 1968. Evolution and adaptive radiation in the Chydoridae (Crustacea:Cladocera): a study in comparative functional morphology and ecology. Philosophical Transactions of the Royal Society of London 254:221-385.

Gannon, J.E. and R.S. Stemberger. 1978. Zooplankton (especially crustaceans and rotifers) as indicators of water quality. Transactions of American Microscopical Society 97:16-35.

Gasith, A. and M.V. Hoyer. 1998. Structuring role of macrophytes in lakes: changing influence along lake size and depth gradients. Pages 381-389 *in* Jeppesen, E., Søndergaard, M., Christoffersen K. (editors), The structuring role of submerged macrophytes in lakes. Springer, New York.

Gotceitas, V. and P. Colgan. 1989. Predator foraging success and habitat complexity: quantitative test of the threshold hypothesis. Oecologia 80:158-166.

Jeppesen, E., J.P. Jensen, P. Kristensen, M. Søndergaard, E. Mortensen, O. Sortkjaer and K. Olrik. 1990. Fish manipulation as a lake restoration tool in shallow, eutrophic, temperate lakes 2: Threshold levels, long-term stability and conclusions. Hydrobiologia 200/201: 219-227.

Lemly, A.D. and J.F. Dimmick. 1982. Growth of young-of-the-year and yearling centrarchids in relation to zooplankton in the littoral zone of lakes. Copia 2:305-321.

Mehner, T. and R. Thiel. 1999. A review of predation impact by 0+ fish on zooplankton in fresh and brackish waters of the temperate northern hemisphere. Environmental Biology of Fishes 56:169-181.

Meijer, M.L., E. Jeppesen, E. van Donk., B. Moss, M. Scheffer, E. Lammens, E. Van Nes, J. A. Berkum, G. J. de Jong, B. A. Faafeng, and J. P. Jensen, 1994. Long-term responses to fish-stock reduction in small shallow lakes: interpretation of five year results of four biomanipulation cases in the Netherlands and Denmark. Hydrobiologia 275/276:457-466.

Michaletz, P.H., D.G. Unkenholz, and C.C. Stone. 1987. Prey size selectivity and food partitioning among zooplanktivorous age-0 fishes in Lake Francis Case, South Dakota. American Midland Naturalist 117:126-138.

Mittelbach, G.G. 1981. Patterns of invertebrate size and abundance in aquatic habitats. Canadian Journal of Fisheries and Aquatic Sciences 38:896-904.

Mittelbach, G.G. 1988. Competition among refuging sunfishes and effects of fish density on littoral zone invertebrates. Ecology 69:614-623.

Moss, B., S. McGowan, and L. Carvalho. 1994. Determination of phytoplankton crops by top-down and bottom-up mechanisms in a group of English lakes, the West Midland Meres. Limnology and Oceanography 39:1020-1029.

Paterson, M. 1993. The distribution of microcrustacea in the littoral zone of a freshwater lake. Hydrobiologia 263:173-183.

Pennak, R.W. 1966. Structure of zooplankton populations in the littoral macrophyte zone of some Colorado lakes. Transactions of the American Microscopical Society 85:329-349.

Quist, M.C., L. Bruce, K. Bogenschutz, and J.E. Morris. 2007. Sample size requirements for estimating species richness of aquatic vegetation in Iowa lakes. Journal of Freshwater Ecology 22: 477-492.

Savino, J.F. and R.A. Stein. 1982. Predator-prey interaction between largemouth bass and bluegills as influenced by simulated, submerged vegetation. Transactions of the American Fisheries Society 111: 255-266.

Scheffer, M. 2004. Ecology of shallow lakes. Chapman and Hall, London.

Scheffer, M., S.H. Hosper, M.L. Meijer, B. Moss, and E. Jeppesen, 1993. Alternative equilibria in shallow lakes. Trends in ecology and evolution(TREE) 8:275-279.

Schneider, J.C. 1999. Dynamics of quality bluegill populations in two Michigan lakes with dense vegetation. North American Journal of Fisheries Management 19:97-109.

Schriver, P., J. Bøgestrand, E. Jeppesen, and M. Søndergaard. 1995 Impact of submerged macrophytes on fish-zooplankton-phytoplankton interactions: large-scale enclosure experiments in a shallow eutrophic lake. Freshwater Biology 33:255-270.

Sommer, U. and H. Stibor. 2002. Copepoda-Cladocera-Tunicata: the role of three major mesozooplankton groups in pelagic food webs. Ecological Research 17:161-174.

Stansfield, J.H., M.R. Perrow, L.D. Tench, A.J.D. Jowitt, and A.A.L. Taylor. 1997. Submerged macrophytes as refuges for grazing Cladocera against fish (-3pt) predation: observations on seasonal changes in relation to macrophyte cover and predation pressure. Hydrobiologia 342/343:229-240.

Stirling, G. 1995. *Daphnia* behaviour as a bioassay of fish presence or predation. Functional Ecology 9:778-784.

Strauss, R. E. 1979. Reliability estimates for Ivlev's electivity index, the forage ratio, and a proposed linear index of food selection. Transactions of the American Fisheries Society 108:344-352.

Timms R. M. and B. Moss, 1984. Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in shallow wetland ecosystem. Limnology and Oceanography 29:472-486.

van Donk, E., R.D. Gulati, and M.P. Grimm. 1989. Food-web manipulation in Lake Zwemlust: positive and negative effects during the first two years. Hydrobiological Bulletin 23:19-35.

van Donk, E. and W.J. van de Bund. 2002. Impact of submerged macrophytes including charophytes on phyto- and zooplankton communities: allelopathy versus other mechanisms. Aquatic Botany 72:261-274.

Weider, L.J. and J. Pijanowska. 1993. Plasticity of *Daphnia* life histories in response to chemical cues from predators. Oikos 67:385-392.

Werner, E.E. ,J.F. Gilliam, D.J. Hall, and G.G. Mittelbach. 1983. An experimental test of the effects of predation risk on habitat use in fish. Ecology 64:1540-1548.

Werner, E.E. and D.J. Hall. 1988. Ontogenetic habitat shifts in bluegill: the foraging rate-predation rate trade-off. Ecology 69:1352-1366.

Wiley, M.J., R.W. Gorden, S.W. Waite, and T. Powless. 1984. The relationship between aquatic macrophytes and sport fish production in Illinois ponds: A simple model. North American Journal of Fisheries Management 4:111-119.

Yin, Y., J.S. Winkelman and H.A. Langrehr. 2000. Long Term Resource Monitoring Program procedures: Aquatic vegetation monitoring. U.S. Geological Survey, Upper Midwest Environmental Sciences Center, La Crosse, WI, USA, LTRMP 95-P002-7.

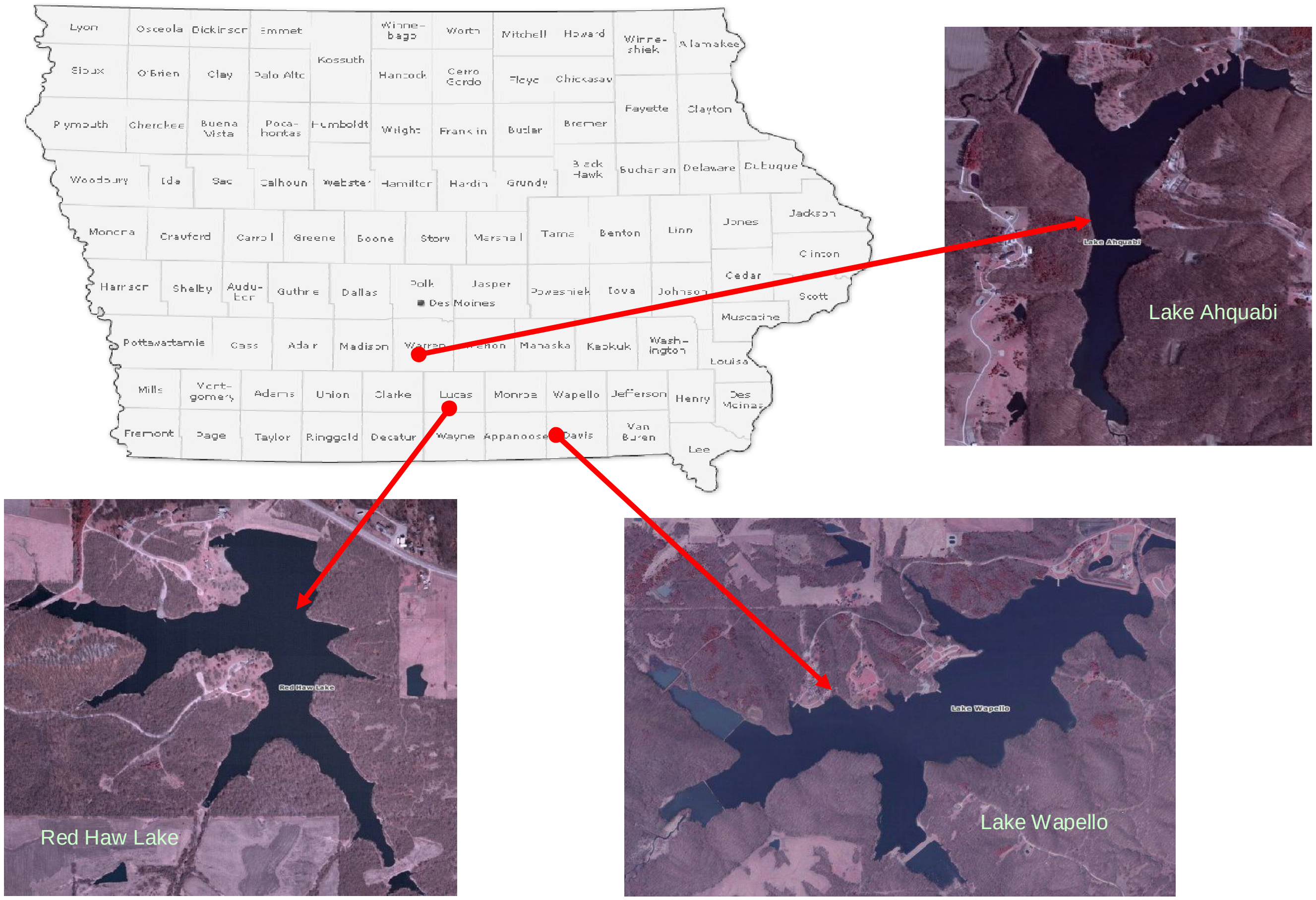

Figure 1. Location and aerial overview of three Iowa study lakes (e.g., Lake Ahquabi, Red Haw Lake, and Lake Wapello).

Table 1. Summary information for the three Iowa study lakes: lake, county, mean depth (m), lake size(ha), and density of grass carp (fish/ha).

Lake	**County**	**Mean Depth (m)**	**Lake size (ha)**	**Grass Carp density (fish/ha)**
Lake Ahquabi	Warren	2.99	47.29	None
Red Haw Lake	Lucas	4.44	30.63	None
Lake Wapello	Davis	3.94	114.33	1.6

Table 2. Mean ± SE statistics for environmental parameters measured during the sampling of emergent/floating and submerged aquatic vegetation in three Iowa lakes from May 2007 to September 2007.

Lake	**Water Temperature (^{o}C)**	**pH**	**Secchi Depth (cm)**	**Total Phosphorus (mg/L) (TP)**	**Total Kjeldahl Nitrogen (mg/L) (TKN)**	**TKN:TP**	**Total Suspended Solids (mg/L)**	**Chlorophyll *a* (ug/L)**	**Alkalinity (mg/L as $CaCO_3$)**	**Hardness (mg/L as CaCO3)**
Ahquabi	24.5 ± 0.3	8.4 ± 0.08	104 ± 24	0.025 ± 0.003	1.244 ± 0.185	48.696	8 ± 2	50 ± 17	88 ± 5	95 ± 7
Red Haw	24.1 ± 0.4	8.1 ± 0.05	145 ± 19	0.016 ± 0.002	0.889 ± 0.070	54.545	5 ± 1	11 ± 2	85 ± 3	90 ± 2
Wapello	25.0 ± 0.2	8.5 ± 0.05	102 ± 10	0.020 ± 0.003	1.122 ± 0.104	55.091	7 ± 1	27 ± 10	85 ± 2	99 ± 6

Table 3. Mean ± SE seasonal zooplankton densities in three Iowa lakes in 2007. Samples were collected by a water pump in the limnetic zone, non-vegetated littoral zone (Open), and vegetated littoral zone (Veg). In addition, a vegetated littoral sample was collected using a box sampler (Box). Levels of densities less than one individual per/liter is labeled as trace (tr).

Lakes	Ahquabi				Red Haw				Wapello			
Submerged Aquatic Vegetation Abundance	6.91±0.225%				15.28±0.595%				7.56±0.186%			
Emerged/Floating Aquatic Vegetation Abundance	4.75±0.259%				9.44±0.319%				19.66±0.431%			
Sample Type	Limnetic	Box	Open	Veg	Limnetic	Box	Open	Veg	Limnetic	Box	Open	Veg
# of Samples Pulled	8	9	29	10	8	12	24	20	8	12	30	22
	# of zooplankton/Liter				# of zooplankton/Liter				# of zooplankton/Liter			
Cladocera	**15 ± 10**	**80 ± 50**	**8 ± 2**	**39 ± 26**	**8 ± 3**	**311 ± 282**	**9 ± 3**	**42 ± 3**	**33 ± 31**	**47 ± 24**	**2 ± 1**	**5 ± 2**
Alona spp.	tr ± tr	8 ± 5	2 ± tr	13 ± 10	1 ± 1	258 ± 198	4 ± 1	93 ± 1	tr ± tr	142 ± 117	1 ± tr	6 ± 2
Alonella spp.	tr ± tr	10 ± 5	tr ± tr	5 ± 4	tr ± tr	57 ± 43	1 ± tr	16 ± tr	tr ± tr	8 ± 5	tr ± tr	1 ± tr
Bosmina spp.	1 ± 1	13 ± 11	8 ± 3	8 ± 5	1 ± 1	42 ± 33	21 ± 10	7 ± 10	60 ± 60	68 ± 38	3 ± 2	8 ± 4
Ceriodaphnia spp.	0 ± ---	7 ± 7	1 ± 1	4 ± 4	0 ± ---	4 ± 4	0 ± ---	5 ± tr	0 ± ---	0 ± ---	tr ± tr	tr ± tr
Chydorus spp.	2 ± 2	135 ± 67	16 ± 6	87 ± 48	0 ± ---	898 ± 892	7 ± 6	120 ± 6	tr ± tr	10 ± 2	2 ± 2	5 ± 2
Diaphanasoma spp.	10 ± 10	1 ± 1	3 ± 1	1 ± tr	4 ± 2	2 ± 2	1 ± tr	tr ± tr	4 ± 3	7 ± 7	tr ± tr	tr ± tr
Daphnia spp.	7 ± 7	0 ± ---	2 ± 1	1 ± 1	13 ± 9	1 ± 1	2 ± 1	1 ± 1	0 ± ---	1 ± 1	2 ± 2	0 ± ---
Scapholeberis spp.	tr ± tr	0 ± ---	3 ± 3	1 ± 1	0 ± ---	0 ± ---	tr ± tr	tr ± tr	0 ± ---	0 ± ---	tr ± tr	1 ± 1
Copepoda	**24 ± 6**	**152 ±** 54	**34 ± 9**	**75 ± 23**	**27 ± 5**	**441 ± 180**	**26 ± 4**	**118 ± 4**	**30 ± 7**	**125 ± 86**	**19 ± 5**	**39 ± 7**
Calanoida spp.	1 ± 1	tr ± 0	tr ± tr	1 ± 1	1 ± tr	0 ± ---	1 ± tr	tr ± tr	7 ± 3	6 ± 5	1 ± 1	tr ± tr
Cyclopoida spp.	16 ± 11	94 ± 47	25 ± 9	95 ± 37	7 ± 3	236 ± 79	18 ± 3	73 ± 3	2 ± 1	75 ± 53	5 ± 1	22 ± 1
Nauplii	41 ± 9	212 ± 70	45 ± 10	73 ± 28	53 ± 11	655 ± 300	45 ± 9	179 ± 9	56 ± 13	186 ± 116	38 ± 14	60 ± 13
Rotifera	**53 ± 24**	**132 ±** 49	**33 ± 9**	**30 ± 10**	**36 ± 14**	**162 ± 69**	**12 ± 3**	**28 ± 3**	**48 ± 11**	**70 ± 40**	**11 ± 4**	**12 ± 3**
Ostracoda	1 ± tr	277 ± 157	12 ± 3	233 ± 165	0 ± ---	238 ± 180	3 ± 2	37 ± 2	4 ± 2	49 ± 23	5 ± 3	15 ± 4

Table 4. Summary of one-way analysis of variance comparing zooplankton samples collected in the limentic, littoral vegetated (Veg), littoral non-vegetated (Open), and littoral vegetated box sample (Box) in three Iowa lakes in 2007. Significant relationship is determined by Bonferroni corrected P-value ≤0.003 (bold).

	Ahquabi				**Red Haw**				**Wapello**			
	F-Ratio	**P-Value**	**Sample Type**	**DF**	**F-Ratio**	**P-Value**	**Sample Type**	**DF**	**F-Ratio**	**P-Value**	**Sample Type**	**DF**
Cladocera	3.36	0.03	-----	3,52	9.37	**<0.0001**	Box&Veg vs. Open&Limnetic	3,60	17.49	**<0.0001**	Box vs. Open	3,68
Alona spp.	5.83	**0.0016**	Box&Veg vs. Open&Limnetic	3,52	18.78	**<0.0001**	Box&Veg vs. Open&Limnetic	3,60	19.20	**<0.0001**	Box vs. Veg vs. Open	3,68
Alonella spp.	8.85	**<0.0001**	Box&Veg vs. Open&Limnetic	3,52	5.30	**0.0026**	Box&Veg vs. Open	3,60	1.31	0.28	-----	3,68
Bosmina spp.	1.58	0.20	-----	3,52	1.28	0.29	-----	3,60	2.28	0.09	-----	3,68
Ceriodaphnia spp.	0.45	0.72	-----	3,52	0.88	0.46	-----	3,60	0.26	0.86	-----	3,68
Chydorus spp.	3.05	0.04	-----	3,52	7.26	**0.0003**	Box&Veg vs. Open&Limnetic	3,60	3.63	0.02	-----	3,68
Diaphanasoma spp.	0.19	0.90	-----	3,52	3.66	0.02	-----	3,60	1.73	0.17	-----	3,68
Daphnia spp.	0.62	0.61	-----	3,52	4.67	0.01	-----	3,60	0.19	0.90	-----	3,68
Scapholeberis spp.	0.26	0.86	-----	3,52	0.67	0.57	-----	3,60	1.86	0.15	-----	3,68
Copepoda	2.32	0.09	-----	3,52	19.79	**<0.0001**	Veg vs. Box vs. Open&Limnetic	3,60	7.00	**0.0004**	Box vs. Open	3,68
Calanoida sp.	2.57	0.06	-----	3,52	0.81	0.50	-----	3,60	4.66	0.01	-----	3,68
Cyclopoida sp.	3.83	0.01	-----	3,52	22.16	**<0.0001**	Box&Veg vs. Open&Limnetic	3,60	16.26	**<0.0001**	Box&Veg vs. Open&Limnetic	3,68
Nauplii	1.00	0.40	-----	3,52	14.03	**<0.0001**	Box vs. Open	3,60	5.31	**0.00**	Box vs. Open	3,68
Rotifera	0.80	0.50	-----	3,52	13.42	**<0.0001**	Box vs. Veg&Limnetic&Open	3,60	11.99	**<0.0001**	Box vs. Open	3,68
Ostracoda	16.09	**<0.0001**	Box&Veg vs. Open&Limnetic	3,52	20.79	**<0.0001**	Box&Veg vs. Open&Limnetic	3,60	12.71	**<0.0001**	Box vs. Open	3,68

Table 5. Total stomach contents of juvenile (≤50 mm) bluegills from spring/summer and fall electrofishing at three Iowa lakes in 2007.

Lakes	Ahquabi						Red Haw						Wapello					
Sample Date	7.13.2007			9.27.2007			7.11.2007			9.28.2007			7.10.2007			9.23.2007		
Sample Size (Fish)	11			15			15			12			6			14		
Total Length (mm)	38-50			36-50			31-50			25-49			36-48			23-49		
Empty Stomachs	1			3			0			0			0			1		
Average Length (mr	43.6 ± 1.3			42.6 ± 1.1			40.1 ± 1.5			34.3 ± 2.6			43.3 ± 1.8			39.3 ± 2.3		
Average weight (g)	1.1 ± 0.1			0.9 ± 0.1			0.8 ± 0.1			0.4 ± .1			1.2 ± 0.1			0.8 ± .1		
Cladocera	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**
Alona sp.	2	10	<1	62	50	9	68	93	19	149	92	25	7	50	2	158	100	63
Alonella sp.	1	10	<1	35	50	5	33	47	9	58	75	10	5	50	1	0	---	---
Bosmina sp.	0	---	---	0	---	---	0	---	---	0	---	---	250	50	63	1	8	<1
Ceriodaphnia sp.	0	---	---	0	---	---	0	---	---	2	8	<1	0	---	---	0	---	---
Chydorus sp.	663	90	64	446	92	64	38	67	11	209	100	35	1	17	<1	69	62	27
Diaphanasoma sp.	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Daphnia sp.	0	---	---	0	---	---	0	---	---	7	50	1	4	33	1	1	8	0
Scapholeberis sp.	0	---	---	0	---	---	1	---	<1	0	---	---	0	---	---	3	23	1
Copepoda																		
Calanoida sp.	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	1	8	<1
Cyclopoida sp.	20	60	2	116	58	17	39	73	11	26	75	4	15	83	4	20	38	8
Nauplii	0	---	---	1	8	<1	0	---	---	5	25	1	0	---	---	0	---	---
Rotifera																		
Asplanchna sp.	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	1	8	<1
Brachionus sp.	0	---	---	0	---	---	0	---	---	2	8	<1	0	---	---	0	---	---
Keratella sp.	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Lecane sp.	0	---	---	0	---	---	0	---	---	3	17	1	0	---	---	0	---	---
Platyias sp.	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Amphipoda	3	20	<1	5	25	1	31	47	9	5	17	1	2	17	1	3	23	1
Insecta																		
Ceratopogonidae	13	30	1	0	---	---	13	40	4	2	8	<1	7	67	2	4	23	2
Chaoboridae	99	10	10	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Chironomidae	24	10	2	4	33	1	124	93	35	93	100	16	105	100	26	52	54	21
Coleoptera	0	---	---	0	---	---	1	7	<1	0	---	---	0	---	---	0	---	---
Diptera larvae	0	---	---	1	8	<1	0	---	---	16	33	3	0	---	---	0	---	---
Diptera adult	0	---	---	0	---	---	1	7	<1	0	---	---	0	---	---	0	---	---
Ephemeroptera	0	---	---	0	---	---	1	7	<1	0	---	---	0	---	---	1	8	<1
Hemiptera	0	---	---	0	---	---	1	7	<1	0	---	---	0	---	---	0	---	---
Lepadella	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Leptoceridae	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Tricoptera	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Odonata	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Ostracoda	206	70	20	28	50	4	5	27	1	13	58	2	2	33	1	39	62	15
Other	0	---	---	1	8	---	0	---	---	0	---	---	0	---	---	0	---	---

Table 6. Total stomach contents of juvenile (>50 mm) bluegills from spring/summer and fall electrofishing at three Iowa lakes in 2007.

Lakes	Ahquabi						Red Haw						Wapello					
Sample Date	Spring/Summer			Fall			Spring/Summer			Fall			Spring/Summer			Fall		
Sample Size (Fish)	20			21			19			16			24			17		
Total Length (mm)	51-114			51-94			51-93			51-83			51-91			52-90		
Empty Stomachs	2			1			1			0			2			2		
Average Length (mm)	69.9 ± 3.2			64.6 ± 2.6			64.1 ± 2.4			61.3 ± 2.2			66.9 ± 2.3			66.3 ± 3.0		
Average weight (g)	6.2 ± 1.3			4.5 ± 0.8			4.2 ± 0.6			2.9 ± 0.4			5 ± 0.6			4.1 ± 0.7		
Cladocera	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**	**Total**	**(%-O)**	**(%-N)**
Alona sp.	9	22	3	121	60	5	20	39	7	78	56	13	3	18	<1	81	53	20
Alonella sp.	20	11	6	57	35	3	7	6	3	30	19	5	1	9	<1	5	13	1
Bosmina sp.	0	---	---	0	---	---	0	---	---	0	---	---	462	14	38	1	7	<1
Ceriodaphnia sp.	0	---	---	0	---	---	0	---	---	2	6	<1	0	---	---	0	---	---
Chydorus sp.	45	33	13	1768	70	78	5	11	2	143	69	23	3	14	<1	53	47	13
Diaphanasoma sp.	0	---	---	5	15	<1	0	---	---	0	---	---	0	---	---	2	13	<1
Daphnia sp.	2	11	1	4	10	<1	1	6	<1	6	19	1	27	18	2	0	---	---
Scapholeberis sp.	0	---	---	1	5	<1	0	---	---	0	---	---	0	---	---	1	7	<1
Copepoda																		
Calanoida sp.	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Cyclopoida sp.	21	39	6	90	80	4	20	44	7	41	50	7	55	32	5	31	53	8
Nauplii	0	---	---	7	15	<1	2	6	1	1	6	<1	0	---	---	0	---	---
Rotifera																		
Asplanchna sp.	1	6	---	0	---	---	0	---	---	2	13	<1	0	---	---	0	---	---
Brachionus sp.	0	---	---	1	5	<1	0	---	---	0	---	---	0	---	---	1	7	<1
Keratella sp.	0	---	---	1	5	<1	0	---	---	0	---	---	0	---	---	0	---	---
Lecane sp.	0	---	---	1	5	<1	0	---	---	0	---	---	0	---	---	0	---	---
Platyias sp.	0	---	---	0	---	---	0	---	---	1	6	0	0	---	---	0	---	---
Amphipoda	4	11	1	3	15	<1	38	39	14	10	31	2	19	32	2	1	7	<1
Insecta																		
Ceratopogonidae	6	11	2	0	---	---	15	28	6	2	6	<1	15	41	1	74	67	18
Chaoboridae	0	---	---	3	5	<1	2	6	1	0	---	---	1	9	<1	0	---	---
Chironomidae	14	33	4	49	60	2	94	78	35	213	88	34	595	100	49	104	87	25
Coleoptera	0	---	---	0	---	---	1	6	<1	0	---	---	0	---	---	0	---	---
Diptera larvae	0	---	---	2	5	<1	10	6	4	48	25	8	0	---	---	0	---	---
Diptera adult	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---	0	---	---
Ephemeroptera	0	---	---	2	10	<1	8	11	3	0	---	---	0	---	---	2	7	<1
Hemiptera	0	---	---	0	---	---	4	11	1	0	---	---	0	---	---	0	---	---
Lepadella	0	---	---	1	5	<1	0	---	---	0	---	---	0	---	---	0	---	---
Leptoceridae	0	---	---	0	---	---	2	11	1	0	---	---	0	---	---	0	---	---
Tricoptera	0	---	---	1	5	<1	0	---	---	0	---	---	1	9	---	0	---	---
Odonata	0	---	---	0	---	0	1	6	<1	0	---	---	0	---	---	0	---	---
Ostracoda	271	17	80	137	55	6	35	56	13	41	56	7	37	45	3	49	60	12
Other	6	6	2	2	10	<1	3	6	1	0	---	0	1	5	0	3	7	1

Table 7. Mean ± SE (P-value) linear indices[1] of food selection of juvenile (≤50 mm) bluegill from spring and fall electrofishing at Ahquabi, Red Haw, and Wapello, Iowa, 2007. Significant relationship is determined by a P-value ≤.05 (bold).

	Ahquabi		Red Haw		Wapello	
Prey	Spring/Summer	Fall	Spring/Summer	Fall	Spring/Summer	Fall
Cladocera						
Alona spp.	0	0.03 ± 0.013 (0.12)	0.07 ± 0.017 **(<0.01)**	0.15 ± 0.039 **(<0.01)**	<0.01 ± 0.015 (1.00)	0.15 ± 0.035 **(<0.01)**
Alonella spp.	-0.03 ± 0.004 **(<0.01)**	-0.02 ± 0.01 **(0.02)**	-0.06 ± 0.021 **(0.02)**	0.01 ± 0.017 (1.00)	-0.05 ± 0.022 (0.12)	<-0.01 ± 0.003 (1.00)
Bosmina spp.	0	<-0.01 ± <0.001 (0.50)	0	0	0.14 ± 0.137 (0.25)	<0.01 ± 0.001 (1.00)
Ceriodaphnia spp.	-0.07 ± 0.0 **(<0.01)**	-0.01 ± 0.005 **(0.01)**	-0.09 ± 0.011 **(<0.01)**	-0.06 ± 0.003 **(<0.01)**	-0.02 ± 0.007 **(0.03)**	-0.03 ± 0.007 **(<0.01)**
Chydorus spp.	0.25 ± 0.09 (0.06)	0.16 ± 0.056 (0.06)	0.05 ± 0.014 (0.09)	0.2 ± 0.057 **(<0.01)**	<-0.01 ± 0.004 (0.62)	0.06 ± 0.022 **(0.04)**
Diaphanosoma spp.	-0.01 ± 0.01 (0.25)	<-0.01 ± <0.01 **(0.03)**	-0.01 ± 0.002 **(<0.01)**	<-0.01 ± 0.001 **(0.01)**	-0.01 ± 0.005 **(0.03)**	<-0.01 ± 0.001 **(<0.01)**
Daphnia spp.	0	0	0	<0.01 ± 0.003 **(0.03)**	0.01 ± 0.006 (0.50)	<0.01 ± 0.001 (1.00)
Scapholeberis spp.	0	0	<-0.01 ± 0.001 (0.22)	<-0.01 ± 0.001 **(0.03)**	<-0.01 ± 0.002 (0.50)	-0.01 ± 0.005 **(0.04)**
Copepoda						
Calanoida	-0.06 ± 0.015 **(<0.01)**	-0.18 ± 0.012 **(<0.01)**	0.02 ± 0.006 **(<0.01)**	-0.14 ± 0.006 **(<0.01)**	-0.12 ± 0.004 **(0.03)**	-0.12 ± 0.006 **(<0.01)**
Cyclopoida	-0.31 ± 0.010 **(<0.01)**	-0.25 ± 0.028 **(<0.01)**	-0.32 ± 0.010 **(<0.01)**	-0.40 ± 0.014 **(<0.01)**	-0.33 ± 0.026 **(0.03)**	-0.37 ± 0.019 **(<0.01)**
Nauplii	-0.32 ± 0.006 **(<0.01)**	-0.44 ± 0.027 **(<0.01)**	-0.16 ± 0.017 **(<0.01)**	-0.18 ± 0.013 **(<0.01)**	-0.25 ± 0.014 **(0.03)**	-0.33 ± 0.023 **(<0.01)**
Rotifera	-0.15 ± 0.015 **(<0.01)**	-0.12 ± 0.013 **(<0.01)**	-0.03 ± 0.002 **(<0.01)**	-0.03 ± 0.004 **(0.01)**	-0.12 ± 0.013 **(0.03)**	-0.03 ± 0.006 **(<0.01)**
Ostracoda	0.15 ± 0.048 **(0.02)**	-0.01 ± 0.014 (1.00)	-0.06 ± 0.017 **(<0.01)**	-0.14 ± 0.039 **(<0.01)**	<-0.01 ± 0.015 (0.69)	-0.11 ± 0.039 (0.04)

[1]Strauss (1979)

Table 8. Mean ± SE (P-value) linear indices[1] of food selection of juvenile (≥50 mm) bluegill from spring and fall electrofishing at Ahquabi, Red Haw, and Wapello, Iowa, 2007. Significant relationship is determined by a P-value ≤.05 (bold).

	Ahquabi		Red Haw		Wapello	
Prey	Spring/Summer	Fall	Spring/Summer	Fall	Spring/Summer	Fall
Cladocera						
Alona spp.	<0.01 ± 0.0036 (0.12)	0.03 ± 0.013 **(0.01)**	<0.01 ± 0.006 (0.17)	0.04 ± 0.017 (0.27)	-0.01 ± 0.003 **(0.01)**	0.05 ± 0.023 (0.39)
Alonella spp.	-0.03 ± 0.007 **(<0.01)**	-0.02 ± 0.009 (0.10)	-0.09 ± 0.013 **(<0.01)**	<-0.01 ± 0.015 (0.23)	-0.05 ± 0.009 **(<0.01)**	-0.02 ± 0.005 **(<0.01)**
Bosmina spp.	0	<-0.01 ± <0.01 (0.06)	0	0	0.04 ± 0.036 (0.50)	<0.01 ± 0.001 (1.00)
Ceriodaphnia spp.	-0.13 ± 0.009 **(<0.01)**	-0.03 ± 0.004 **(<0.01)**	-0.08 ± 0.009 **(<0.01)**	-0.05 ± 0.003 **(<0.01)**	-0.03 ± 0.005 **(<0.01)**	-0.01 ± 0.004 **(<0.01)**
Chydorus spp.	0.01 ± 0.015 (0.06)	0.23 ± 0.070 **(0.05)**	<-0.01 ± 0.003 **(0.01)**	0.09 ± 0.027 (0.12)	<-0.01 ± 0.002 (0.34)	0.03 ± 0.014 (0.34)
Diaphanosoma spp.	-0.02 ± <0.01 **(<0.01)**	<-0.01 ± 0.003 (0.09)	-0.02 ± 0.002 **(<0.01)**	<-0.01 ± 0.001 (0.12)	-0.02 ± 0.003 **(<0.01)**	<-0.01 ± 0.001 (0.07)
Daphnia	-0.03 ± 0.008 **(0.01)**	<0.01 ± 0.001 **(0.05)**	<0.01 ± <0.001 (1.00)	<0.01 ± 0.003 (0.25)	0.01 ± 0.007 (0.74)	0
Scapholeberis spp.	0.01 ± <0.01 **(<0.01)**	0	<-0.01 ± 0.001 (0.06)	<-0.01 ± 0.001 (0.06)	<-0.01 ± 0.001 **(<0.01)**	<-0.01 ± 0.002 (0.50)
Copepoda						
Calanoida	-0.09 ± 0.009 **(<0.01)**	-0.17 ± 0.010 **(<0.01)**	-0.17 ± 0.005 **(<0.01**)	-0.14 ± 0.004 **(<0.01)**	-0.12 ± 0.002 **(<0.01)**	-0.14 ± 0.003 **(<0.01)**
Cyclopoida	-0.22 ± 0.015 **(<0.01)**	-0.30 ± 0.021 **(<0.01)**	-0.35 ± 0.009 **(<0.01)**	-0.41 ± 0.010 **(<0.01)**	-0.35 ± 0.014 **(<0.01)**	-0.39 ± 0.024 **(<0.01)**
Nauplii	-0.33 ± 0.010 **(<0.01)**	-0.37 ± 0.021 **(<0.01)**	-017 ± 0.015 **(<0.01)**	-0.17 ± 0.01 **(<0.01)**	-0.23 ± 0.008 **(<0.01)**	-0.31 ± 0.026 **(<0.01)**
Rotifera	-0.11 ± 0.010 **(<0.01)**	-0.11 ± 0.0125 **(<0.01)**	-0.03 ± 0.002 **(<0.01)**	-0.03 ± <0.01 **(<0.01)**	-0.13 ± 0.008 **(<0.01)**	-0.06 ± 0.004 **(<0.01)**
Ostracoda	0.11 ± 0.042 **(<0.01)**	<0.01 ± 0.022 (1.00)	0.02 ± 0.011 **(0.03)**	-0.01 ± 0.019 (0.795)	0.03 ± 0.008 **(<0.01)**	-0.01 ± 0.025 (0.27)

[1]Strauss (1979)

CHAPTER 4. GENERAL CONCLUSIONS

Aquatic vegetation plays a vital role in maintaining the overall integrity of aquatic ecosystems (i.e., lakes, ponds, streams, and rivers). It stabilizes aquatic ecosystems by lowering nutrient concentrations (van Donk et al. 1989), increasing water clarity, producing oxygen, reducing shore erosion, and providing food and habitat for aquatic fauna (Canfield et al. 1984; Timms and Moss 1984; Jeppesen et al. 1990; Scheffer et al. 1993; Meijer et al. 1994; Moss et al. 1994; Egertson et al. 2004). Aquatic vegetation is influenced by factors such as irradiance, temperature, wave action, lake size, catchment basin morphology, and water chemistry (Gasith and Hoyer 1998).

Many studies have shown the positive correlation between maximum depth for vegetation growth and water clarity (Canfield et al. 1985; Chambers and Kalff 1985). This correlation is supported by research garnered from my research in that submerged aquatic vegetation abundance was positively related to water clarity (Secchi-depth) in the 13 study lakes. Scheffer (2004) explains in the absence of aquatic vegetation, chlorophyll *a* increases with a rise in TP levels. My study lakes followed this trend in that the increase of chlorophyll *a* and TP levels is likely due to increased lake turbidity and phytoplankton abundance. Submerged aquatic vegetation, in turn, has limited growth potential. Canfield et al. (1984) showed that with increased volume of aquatic vegetation in a lake, chlorophyll *a* concentrations decrease. Finding similar results, the lake with the highest aquatic vegetation (Red Haw, abundance= 15 ± 0.6 %) had the lowest chlorophyll *a* concentrations (11 ± 2 µg/L) as well as the lowest TP average (0.016±0.002mg/L).

In addition to the direct relationship between specific water quality parameters to aquatic vegetation densities, ordination reviewed a strong negative relationship between Secchi-depth and chlorophyll a levels, and lakes that share these characteristics. Lakes with little vegetation are exemplified by, high chlorophyll a levels, high TSS, and low Secchi-depths. In contrast, lakes that have more aquatic vegetation abundance and species diversity are noted by having greater Secchi-depths and lower chlorophyll a and TSS levels. My study also revealed the importance of sampling gear and location in sampling zooplankton in lakes. There was a noted similarity between pump samples collected from vegetative area and those collected using a box sampler compared to those collected in the limnetic and non-vegetated littoral locations in the Lakes Ahquabi, Red Haw, and Wapello. Cladocerans, especially from the family of Chydoridae, have been shown to have high population abundance in plant beds (Scheffer 2004). Chydorids have appendages and behaviors that are adapted to living on plant surfaces, and are difficult to dislodge even when disturbed (Pennak 1966; Fryer 1968; Campbell et al. 1985). Lake Red Haw had the largest chydorid population as well as the highest submerged aquatic vegetation abundance consisting primarily of a dense population of *Ceratophyllum demersum* (89%)

Regardless of fish size (≤50mm and >50mm), prey selectivity was similar but fall and spring seasons did influence prey selection. Dewey et al. (1997) determined young bluegill fed on small prey (e.g., chydorids, *Daphnia* spp., and other cladocera, while adult bluegills consumed amphipods, gastropods, and odonates. However diets during both life stages consisted of chironomids (Dewey et al. 1997).

Red Haw Lake, the highest submerged aquatic vegetation and highest water clarity of the study lakes, bluegills (<50mm) fed on chironomids (35% of diet) and amphipods (14% of diet) in spring. During the fall sampling period, bluegills feeding habits shifted from amphipods (2% of diet) to *Chydorus* spp. (23% of diet). Seasonal variation in bluegill diets may have, in part, been related to life history stages of prey (e.g., aquatic insects; Mittelbach 1981). In the fall, larval aquatic insect populations are low because most have matured and emerged as aerial adults (Mittelbach 1981). During this seasonal change, juvenile fish feed primarily on chydorids, gastropods, amphipods, *Daphnia* spp., and *Bosmina* spp. (Dewey et al. 1997). In all three lakes, chironomids as a food source were observed less frequently during the fall sampling season than during the spring/summer sampling season, supporting Mittelbach (1981) and Dewey et al. (1997) findings.

The information acquired from my research can be used for other fish species. To improve recruitment in areas with minimal submerged aquatic vegetation, programs to increase submerged aquatic vegetation should be implemented. In addition, a balance between complete eradication and severe infestation of aquatic vegetation is needed. In our study, lakes with chlorophyll *a* levels around 60 μg/L,TKN levels around 2 mg/L, Secchi-depth near 100 cm and TSS around 10 mg/L appear to be the limit between higher SAV abundance and lower SAV abundance. However, since all of our lakes are impoundments, lakes with natural origins might have different management guidelines. Management efforts should focus on aquatic vegetation densities that optimize lakes resources (e.g., water clarity, sport fish populations, recreational activities).

Recommendation for Future Research

The role of aquatic vegetation has become increasing recognized as being important tool in effective fishery management. Future research should focus on the causative factors in influence aquatic lakes when the lakes that have similar abiotic and biotic conditions still exhibit completely different aquatic vegetation abundance and diversity.

Also, while watershed improvements have been instrumental in improving water quality in lakes, the increased water clarity has often resulted in increased aquatic vegetation infestation. Although aquatic vegetation is important, so is access to a specific water body for other recreational uses beyond fishing. What is needed to better manage both the aquatic vegetation and associated fishery, as well as the lake as a whole?

In addition to whole lake system research, small-scale, well-controlled, and replicated experiments need to be used to help determine how aquatic vegetation can combat turbid, cyanobacteria-dominated lakes back to a clear-water state system. The role of aquatic vegetation and the best means to achieve multiple positive results still need to be investigated. With the continuation of degraded aquatic ecosystems due to anthropogenic effects, finding possible solutions for these damaged systems will ensure a good fishery and recreational fun for years to come.

REFERENCES

Canfield, D.E. Jr., K.A. Langeland, S.B. Linda, and W.T. Haller. 1985. Relations between water transparency and maximum depth of macrophyte colonization in lakes. Journal of Aquatic Plant Management 23:25-28.

Canfield, D. E., J.V. Shireman, D.E. Colle, W. T. Haller, C. E. Watkins, and M. J. Maceina. 1984. Prediction of chlorophyll *a* concentrations in Florida lakes: importance of aquatic macrophytes. Canadian Journal of Fisheries and Aquatic Sciences 41(3):497-501.

Chambers, P.A., and J. Kalff. 1985. Depth distribution and biomass of submersed aquatic macrophyte communities in relation to Secchi depth. Canadian Journal of Fisheries and Aquatic Sciences 42:701–709.

Dewey, M.R., W.B. Richardson, and S.J. Zigler. 1997. Patterns of foraging and distribution of bluegill sunfish in a Mississippi River backwater: influence of macrophytes and predation. Ecology of Freshwater Fish 6:8-15.

Egertson ,C. J., J.A. Kopaska, and J.A. Downing. 2004. A century of change in macrophyte abundance and composition in response to agricultural eutrophication. Hydrobiologia 524:145-156.

Fryer, G. 1968. Evolution and adaptive radiation in the Chydoridae (Crustacea:Cladocera): a study in comparative functional morphology and ecology. Philosophical Transactions of the Royal Society of London 254:221-385.

Gasith, A. and M.V. Hoyer. 1998. Structuring role of macrophytes in lakes: changing influence along lake size and depth gradients. Pages 381-389 *in* Jeppesen, E., Søndergaard, M., Christoffersen K. (editors), The structuring role of submerged macrophytes in lakes. Springer, New York.

Jeppesen, E., J.P. Jensen, P. Kristensen, M. Søndergaard, E. Mortensen, O. Sortkjaer and K. Olrik. 1990. Fish manipulation as a lake restoration tool in shallow, eutrophic, temperate lakes 2: Threshold levels, long-term stability and conclusions. Hydrobiologia 200/201:219-227.

Meijer, M.L., E. Jeppesen, E. van Donk., B. Moss, M. Scheffer, E. Lammens, E. Van Nes, J. A. Berkum, G. J. de Jong, B. A. Faafeng, and J. P. Jensen, 1994. Long-term responses to fish-stock reduction in small shallow lakes: interpretation of five year results of four biomanipulation cases in the Netherlands and Denmark. Hydrobiologia 275/276:457-466.

Mittelbach, G.G. 1981. Patterns of invertebrate size and abundance in aquatic habitats. Canadian Journal of Fisheries and Aquatic Sciences 38:896-904.

Moss, B., S. McGowan and L. Carvalho. 1994. Determination of phytoplankton crops by top-down and bottom-up mechanisms in a group of English lakes, the West Midland Meres. Limnology and Oceanography 39:1020-1029.

Pennak, R.W. 1966. Structure of zooplankton populations in the littoral macrophyte zone of some Colorado lakes. Transactions of the American Microscopical Society 85:329-349.

Scheffer, M., S.H. Hosper, M.L. Meijer, B. Moss, and E. Jeppesen, 1993. Alternative equilibria in shallow lakes. Trends in ecology and evolution(TREE) 8:275-279.

Scheffer, M. 2004. Ecology of Shallow Lakes. Chapman and Hall, London.

Timms R. M. and B. Moss, 1984. Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in shallow wetland ecosystem. Limnology and Oceanography 29:472-486.

van Donk, E., R.D. Gulati, and M.P. Grimm. 1989. Food-web manipulation in Lake Zwemlust: positive and negative effects during the first two years. Hydrobiological Bulletin 23:19-35.

ACKNOWLEDGMENTS

I would like to thank my advisor, Dr. Joseph Morris, for his guiding words and encouragement through this challenging but rewarding experience. Also thank you to my committee members, Drs. Michael Quist and John Downing for your insight and prompt feedback to all my questions. In addition without the funding, initial planning, and steadfast support from the Iowa Department of Natural Resources, in particular, Darcy Cashatt and Lewis Bruce, this project would not have been successful. Thank you Rich Clayton for taking time away from your family to be my technician in year one and fix all the equipment that seemed to break. It is truly appreciated and your work does not go unnoticed. Also I would like to acknowledge Andy Fowler for setting up the database and Garritt Page for spending many hours helping me with analysis.

Then there was the crew, my army, my saving grace. Without them I would still be in the basement counting zooplankton, dissecting stomachs, and entering data. They not only were fantastic technicians but also true friends so a giant thanks to: Aaron Cole, Adam Havard, Bonnie Mulligan, Erin Mugge, Jamie Bozwell, Larissa Havard, Luke Brown, and Tanner Francisco.

I knew I was working on my masters, but I got a surprise; a friendship was born and bond was formed over the triumphs (changing a flat tire), tribulations (getting the flat tire) and life stories shared over lunches at Subway. To this day I have never spent as much time with one person as I did with Abby Mayer, my 2007 technician, and I thank her for one of the best summers. I am forever indebted to

Abby as she stood by my side during the hardest year in my life, making sure that I never finished alone.

Finally, I would like to thank God for the gift he gave me- my friends and family. Their words of encouragement, constant nurture from a young age, and recent support have given me the drive to finish-thank you Mom, Brian, and Jenny. The completion of my thesis is bittersweet without the congratulatory hug from my Dad. His courageous battle ended a little too soon to see me finish, but he is looking down, proud as ever; for that, my master's degree is dedicated to him.